How has woody vegetation changed in north-east Namibia in response to land use, climate and fire?

Conor Eastment

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Supervisors: A/Prof. Lindsey Gillson, Dr. Glynis Humphrey and Prof. Timm Hoffman

Percy FitzPatrick Institute of African Ornithology University of Cape Town, Rondebosch

Cape Town, South Africa 7701

University

University of Cape Town

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ii) Acknowledgements:

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ABSTRACT:

Bush encroachment or the thickening of woody vegetation is a phenomenon occurring throughout savannas, which tends to be more pronounced in small protected areas. The consequences of bush encroachment are often negative for the conservation of biodiversity, for the promotion of tourism and the prevention of wildfires. Hence, effective monitoring of woody vegetation and the factors which influence its spread are essential. This is particularly the case for protected areas such as that of Bwabwata National Park (BNP) in north-east Namibia. With a complex land use history and different fire management approaches being adopted throughout the area, the effect of fire on woody vegetation in BNP is currently poorly understood. This study used a 20-year-old repeat photography monitoring project and satellite-based remote sensing products to explore woody cover dynamics in BNP. Results revealed that woody cover has increased by 13% since 1999 in BNP. Furthermore, the results show differences in the structure of woody vegetation. Repeated late dry season fires in the west of the park have driven an increasing dominance of <3 m woody cover which is unable to escape the 'fire-trap'. A combination of the rainfall gradient from west to east and shorter fire return intervals resulting in less late dry season fires has driven an increase in woody cover >3m in eastern sections of the park. This influence of different fire regimes spatially across BNP, suggests that local fire management is a significant determinant of woody vegetation change. Woody vegetation change differs spatially across BNP due to frequent late dry season fires prevailing in the west and less frequent earlier season fires occurring in the east. Therefore, in order to reduce the mortality of woody species and conserve heterogenous height structure in the west, a reduction of frequent late dry season fires is required. Early dry season fires are shown to reduce the rate of increasing total woody cover change and, therefore, this fire management strategy arguably contributes towards the reduction of wildfire risk, conservation of biodiversity and promotion of tourism.

Key Words: woody vegetation change, repeat photography, fire management

CHAPTER 1: INTRODUCTION

1.1 . Trends in Woody Vegetation Change in southern Africa

Savannas are a globally significant biome, covering one eighth of the planet's surface. They form a large portion of the protected area network of Africa and make up half of the continent's surface (Scholes & Archer 1997). A significant portion of the continent's growing human population, livestock and rangeland occurs within the savanna biome, which is home to high faunal and floral biodiversity (Scholes & Archer 1997; O'Connor & Stevens 2017). Savannas are defined by a continuous grass layer with a discontinuous tree layer (O'Connor & Stevens 2017). The relative proportions of grass compared to trees is the most important relationship for understanding savanna systems (O'Connor & Stevens 2017). Forty percent tree cover is currently understood as the mid ground between a grassy savanna and a woody savanna, which, depending on the mix of shrubs and trees would either be considered as a tree savanna or shrub savanna (Staver et al. 2011; Scholes 2003). Once tree cover reaches 50% the savanna system is classified as savanna woodland (Staver et al. 2011; Scholes 2003).

Across southern Africa and indeed many parts of the globe, a trend of increasing woody vegetation has been observed in savanna and grassland biomes (Archer et al. 2017; O'Connor et al. 2014; O'Connor & Stevens 2017). Within southern African land use systems, consisting of commercial, communal, protected area and mixed use, O'Connor et al., (2014), found that 22 of the 23 studies in their review exhibited an increase in woody vegetation. Of all these land use types, small protected areas were the most vulnerable to bush encroachment as they displayed the highest annual increase in woody vegetation (O'Connor et al. 2014). This trend has been described as bush encroachment, but other terms such as woody thickening and woody plant encroachment, are used in reference to increasing density of indigenous woody plants (Archer et al. 2017; O'Connor & Stevens 2017; O'Connor et al. 2014).

An increase in woody vegetation, depending on its rate and extent, can give rise to a fundamental shift in ecosystem structure. This, in turn, has widespread consequences for the services (e.g. food, water, climate regulation) that ecosystems provide (Eldridge et al. 2011). These consequences can be split into four primary categories of impact which are influenced to some extent by the woody species involved: 1) primary and secondary production, 2) ecosystem functioning, 3) biodiversity and 4) tourism (O'Connor & Stevens 2017; O'Connor et al. 2014; Archer et al. 2017; Eldridge et al. 2011).

The following broad overview of consequences can be considered (Archer et al. 2017; Eldridge et al. 2011):

1) With an increase in woody vegetation, there is a subsequent decrease in herbaceous cover as a result of the ground layer being shaded out (Scholes 2003; Smit 2004; Walker et al. 1972). This reduction in primary production in the herbaceous layer can have multiple effects on secondary production. For example, the loss of grazing resources for grass dependent herbivores found throughout savanna and grassland systems (Smit & Prins 2015; Archer et al. 2017).

2) Alteration of the grass-tree balance can impact the aboveground net primary productivity (ANPP) of a system and in so doing can alter the carbon dynamics of encroached systems (Jackson et al. 2002; Hudak et al. 2003; Knapp et al. 2008). In arid sites, which have been encroached by woody plants, ANPP has been shown to decrease and in moister sites to increase (Knapp et al. 2008). Considering the current importance of monitoring the carbon cycle, any fluctuation in carbon sequestration (such as increased soil carbon as a result of encroached systems) requires attention (Jackson et al. 2002; Hudak et al. 2003; Knapp et al. 2008). While some woody species associated with bush encroachment may be seen to decrease climate warming through the increased uptake of carbon and evaporative cooling (Archer et al. 2001; Bonan 2008), they might simultaneously contribute to the warming trend by decreasing albedo (Bonan 2008).

   Encroachment also impacts catchments as a result of increasing erosion and rainfall interception (Grellier et al. 2012; Honda & Durigan et al. 2016; Huxman et al. 2005; Zou et al. 2014). In a study which evaluated the effect of invasion of grassland by woody species, sediment loss and gulley expansion were positively correlated with encroachment (Grellier et al. 2012). Increasing woody species cover results in higher levels of water uptake through the process of interception which directly reduces streamflow and levels of water in soil (Honda & Durigan et al. 2016; Huxman et al. 2005; Zou et al. 2014). Uptake of water (and nutrients) by increasing cover of woody species translates into a larger biomass than would be found in a savanna with a more mixed tree-grass community (van Wilgen et al. 2014). Increased woody cover shades out and reduces grass cover which subsequently reduces the likelihood of a fire (Hoffmann et al. 2005). However, higher woody vegetation biomass results in an increased risk of wildfires which poses challenges to management, local communities and fire sensitive flora and fauna (van Wilgen et al. 2014).

3) Increasing dominance of woody species has multiple negative impacts on species richness in savanna and Grassland systems (Smit & Prins 2015; Sirami et al. 2009; Ratajczak et al. 2012; Blaum et al. 2007; Baez & Collins et al. 2008). In addition to a reduction of grass and herb productivity, plant species richness and stability of species composition is negatively associated with increasing woody vegetation (Baez & Collins et al. 2008; Ratajczak et al. 2012). Possibly as a knock-on effect of the reduced quality of groundcover habitat, abundance of small carnivores and their prey is reduced with high levels of encroachment (Blaum et al. 2007). A similar reduction in avian species richness has been found as a result of encroachment. However, in the case of birds, the negative impact was as a result of altered vegetation height structure more than level of encroachment or species responsible for encroachment (Sirami et al. 2009).

4) While impacts on tourism can be extrapolated from consequences of encroachment already stated, a reduction in tourism potential as a result of encroachment warrants special mention due to the extremely vital opportunity wildlife tourism can play in many local communities and conservancies (Weaver & Skyer 2003). Encroachment, in some cases has been shown to reduce visibility in conservation areas making wildlife viewing difficult (Gray & Bond 2013). The increased difficulty of viewing would reduce tourist's perception of the conservation area which impacts their willingness to pay park fees. This is a source of income directly linked to a park or conservation area's ability to conserve biodiversity (Gray & Bond 2013).

1.2 . Local vs Global - Drivers of Woody Vegetation Change:

Increase in woody vegetation has traditionally been understood through the processes described by two contrasting models. These are the root-niche or competition-based models and the demographic bottleneck model (O'Connor & Stevens 2017).

Root-niche theory or the competition-based model was first described by Walter (1971), who explains that tree and grass co-existence occurs due to a separation of rooting niches. The idea is that trees and grasses compete for nutrients and water through their rooting systems which are deeper for trees and shallower for grasses (O'Connor et al. 2014). Sandy soils allow more infiltration and favour trees. Conversely, clay rich soils hold more water and favour grasses (Walker & Noy-Meir 1982). This model is classically explained by case studies where heavy grazing has reduced the ability of grass to compete for water, increasing infiltration and favouring tree establishment (Walker et al. 1981; O'Connor et al. 2014).

Demographic bottleneck models focus less on competition for resources than on the role of fire or herbivory disturbances and climatic variability which can limit tree growth and thus allow for the co-existence of grass and trees (O'Connor & Stevens 2017). In seasons of low rainfall, germination of woody species is reduced and thus tree cover is limited (Higgins et al. 2000; Sankaran et al. 2005). High frequency and intensity of fire or dense herbivore numbers increase disturbance and reduce tree cover by limiting the ability of trees to reach maturity. This is affected, in turn by the growth rate of the species involved and by their access to above and below ground resources (Higgins et al. 2000). This model is supported by O'Connor et al. (2014) who reviewed the southern African literature on woody plant encroachment and found that areas with fire suppression policies had higher rates of encroachment. Due to a reduction in the disturbance regime of fire, woody species are able to escape what is known as the 'fire trap' and reach a height which renders them more resilient from further disturbance (Higgins et al. 2000; O'Connor et al. 2014; van Wilgen et al. 2014). The height at which woody species are thought to escape the 'fire trap' is variable. A height greater than 3 m is generally considered enough to escape most fires (Buitenwerf et al. 2012). At this height, only frequent, intense and late season fires are affective at reducing woody cover (Govender et al. 2006; van Wilgen et al. 2014)

The role of carbon dioxide (CO_2) in promoting bush encroachment has been studied and supported owing primarily to the effect it has on the growth of many woody species (Higgins & Scheiter 2012; Haverd et al. 2020). CO_2 decreases stomatal conductance and transpiration rates of woody species facilitating faster growth rates and recovery from disturbance events (Kgope et al. 2010). This growth advantage presented by CO_2 does not have the same effect on grasses and, therefore, globally increasing levels of CO_2 provide a competitive advantage to woody vegetation (Kgope et al. 2010). Additionally, while land use may affect the pace of encroachment, multiple studies have shown that the increase of woody species is occurring across land use types, disturbance regimes and climatic gradients (Buitenwerf et al. 2012; Wigley et al. 2009; O'Connor et al. 2014). This evidence suggests that while local factors emphasised by the competition based and demographic bottleneck models influence encroachment, it is global drivers such as climate change and increasing CO_2 which are primarily responsible for the global phenomenon of woody species encroachment (Higgins & Scheiter 2012; Haverd et al. 2020).

1.3 . Understanding Woody Vegetation Change

The study of woody vegetation change requires multi-disciplinary methods across a range of temporal and spatial scales with varying frames of inquiry (O'Connor & Stevens 2017). There are three primary ways of analysing change. The first is through the use of correlative studies of temporal and spatial patterns of woody vegetation change. The second is by using experiments such as fire exclusion plots which are intended to allow comparison of changes with non-fire excluded areas of similar characteristics (van Wilgen et al. 2014). The third approach is to use process level studies such as an analysis of the physiological effect of CO_2 on woody species vs. grass species (Kgope et al. 2010; Haverd et al. 2020). The spatial and temporal scale of these studies varies depending on the intended application and motivation of study (Gilson 2009; Gillson & Willis 200. Process and experimental based studies provide insight into detailed ways in which change occurs. This can inform specific management decisions on how best to go about adapting and responding to ecological changes. Understanding temporal and spatial patterns of change provides perspective on how these changes are related to thresholds of potential concern (ToC) (*sensu* ToC in Gillson & Duffin 2007; Gillson 2009). Once perspective is gained on how contemporary change relates to changes which occurred in the past and are expected in the future, more detailed management lessons gained from experimental and process level studies may be more appropriately applied (Gillson 2009; Gillson & Duffin 2007).

The primary approaches used to assess the temporal and spatial patterns of change in woody vegetation cover include aerial photography, ground-based repeat photography, satellite-based remote sensing and palaeo methods (O'Connor & Stevens. 2017). These methods provide insight into the changes that have occurred over a relatively large area compared to traditional vegetation monitoring plots (Webb et al. 2010; Paine & Kiser 2012; Chuvieco 2009 O'Connor & Stevens 2017). To compensate for the strengths and weaknesses of these methodologies, they are often combined with on-the-ground vegetation monitoring or other methods which are able to provide a more detailed understanding of the changes that have occurred in vegetation structure and species composition (O'Connor & Stevens 2017; O'Connor et al. 2014). Palaeo methods use multiple proxies to understand centuries to millenia scale changes in vegetation. Two examples of the approaches used are, firstly, the use of charcoal deposits to understand thousand-year-old fire regimes (e.g. Dixon 2019) and secondly the comparison of modern pollen and fossil pollen assemblages to understand vegetation transitions over millennia (e.g. Julier et al. 2018). In comparison with palaeo-based methods, remote sensing and photography provide snapshots in time of vegetation change. Satellite based remote sensing datasets (e.g. Landsat or MODIS) provide large spatial scale and detail, but only within decadal time scales (Chuvieco 2009).

Aerial photography can be used for a temporal scale of less than a century but is limited to areas where flight paths and monitoring exist (Paine & Kiser 2012). Ground-based repeat photography (i.e. fixed point, repeat or 'matched' photography) provides decadal scales of change which is ideal for studying woody vegetation (Webb et al. 2010).

A major strength of ground-based repeat photography is that it provides a detailed understanding of species presence and height structure, which are difficult to determine from aerial photography and satellite-based remote sensing products (Webb et al. 2010). When ground photos are repeated with exactly the same frame during the same time of year as the original photo, woody vegetation change and it's rate can be measured with insight into the species responsible for that change and how height structure has changed over the period of time (Webb et al. 2010; Rohde et al. 2019; Hoffman et al. 2010).

1.4. Local Management and Woody Vegetation Change

Responses to change in woody vegetation depend on the goals and objectives of the authority responsible for management of the land where change has taken place. In the case of the study area of this thesis, the protected area of Bwabwata National Park (BNP) is managed by the Ministry of Environment and Tourism (MET) (MET 2013). The primary objectives of BNP are the preservation of biodiversity and to improve economic development through the principle of sustainable utilisation (MET 2013). BNP occurs within the greater Kavango Zambezi - Transfrontier Conservation Area (KAZA - TFCA) and the movement of game is relatively unrestricted by fences. Therefore, the primary method of controlling woody vegetation is through the use of fire by local management authorities. The objectives of BNP's fire management strategy are to minimise the risk of wildfire, enhance park use and to improve ecosystems and habitats (MET 2013). Thus, some of the goals of BNP may be compromised by the consequences of increased woody vegetation and the efficiency of their fire management strategy is crucial in achieving the stated park objectives.

In order to implement an effective fire management strategy, monitoring of variables which influence the nature of fires is vital (van Wilgen et al. 2014). Thus, at least a baseline understanding of the state of the tree-grass balance and other drivers of woody vegetation change is required. Rainfall is a significant determinant of fire activity due to its influence on grass and tree growth (Archibald et al. 2009; Sankaran et al. 2005). Seasons of high rainfall promote increased levels of grass growth and cover (Archibald et al. 2009; van Wilgen et al. 2014; Sankaran et al. 2005). High grass cover is associated with increased fire activity as most fires in savanna systems start as ground layer grass fires.

Depending on the grass and tree fuel load and their moisture level, such fires can either become large intense fires or small cool fires (Govender et al. 2006). Thus, monitoring of antecedent rainfall (generally the previous 18 months of rainfall (Archibald et al. 2009; Archibald et al. 2010) is important for fire management as it affects grass density (i.e. fuel for fires) and plays a role in the rate and extent of burnt area in savanna woodlands (Archibald et al. 2009; Archibald et al. 2010).

Historically, a policy of fire suppression was adopted by many protected areas including that of BNP (Humphrey 2018). However, fire suppression has subsequently been found to contribute to bush encroachment across many study sites (O'Connor et al. 2014; O'Connor & Stevens 2017). Globally, this has prompted an increased recognition of the important role that fire management plays in managing woody vegetation. A common method used to control woody vegetation is the use of infrequent late dry season (LDS) fires which can burn large amounts of biomass due to their high intensity, large size and high rate of spread (van Wilgen et al. 2014; Govender et al. 2006). In contrast to this approach is that of frequent early dry season (EDS) burning when vegetation is moist, and fires are less intense, smaller and have low spreading rates (Govender et al. 2006; Humphrey 2018). EDS burning is easier to control and promotes mosaic patch burning. This regime is also associated with increased habitat heterogeneity (Brocket et al. 2001) and has been practiced by local communities within BNP for centuries, if not for thousands of years (Humphrey 2018). The traditional use of EDS burning has predominantly taken place in the eastern inhabited area of BNP where fire knowledge has been handed down through the generations and has been recognised by BNP management as an effective strategy (MET 2016; Humphrey 2018).

1.5. Research Question, Aim and Objectives with Hypotheses

Research Question:

How has woody cover changed over the last two decades (1999 – 2019) in BNP and what factors have been responsible for this change?

Bwabwata National Park has evolved in response to a complex set of social and ecological historical events (c. 1700s – present) that have shaped its current environmental and land use context (Humphrey 2018). Effective management of woody cover is vital for the conservation of biodiversity, for preventing wildfires and for encouraging tourism within BNP (MET 2013; MET 2016). This research builds on previous monitoring efforts in the park and is based on a repeat photography approach to vegetation monitoring. It uses a combination of repeat photography and satellite-based remote sensing products for describing and analysing the state of woody cover vegetation within BNP. The primary aim of this study is to understand how fire, land use and rainfall have influenced woody cover change in BNP and how this knowledge can be used to manage the vegetation of the region.

Objectives:

1. To investigate the nature, extent and rate of woody cover change (1999 – 2019) using repeat photography and remote sensing products.

Considering the pattern of increasing woody cover and reduction of large mature trees in savannas across southern Africa (O'Connor et al. 2014), my objective was to explore whether the nature of change within BNP conformed to this trend or not. If BNP did conform to similar study sites in the review of O'Connor et al. (2014), then I hypothesised that the increase in woody vegetation would have occurred at a rate of approximately 0.5 – 0.6% per annum. Total woody cover (defined as canopy cover), height structure and species composition were compared between two disparate land use types that included core conservation areas and inhabited zones in BNP. A greater than 3m height category was used as a proxy for tree cover to investigate any possible transitions from grass to woody savanna states as described by Staver et al. (2011) and Scholes (2003). Woody cover below 3m was considered as shrub or juvenile tree cover. This approach integrated the strengths of a repeat photography study (inclusive of recording height structure and species composition) with the products derived from remote sensing to quantify woody cover change. This ensured a replicable study, that could potentially be useful for monitoring woody cover change for park management purposes in the future .

2. To investigate inter-decadal scale patterns in rainfall, fire and land use at repeat photography sites.

This objective investigated how rainfall, land use and fire interact with and influence woody cover change across BNP. Mean Annual Precipitation (MAP) at each photo site was reconstructed using modelled climate data. The accuracy of the modelled data was tested against observed rainfall. Rainfall from the season of 1996 - 1997 – 2018 - 2019 was calculated in order to investigate antecedent rainfall and the gradient of rainfall from west to east of BNP. Average fire seasonality and frequency for photo sites was extracted using the MODIS burned area product. The land use areas of BNP differ in their management and application of fire (MET 2013; MET 2016), and I hypothesised that there would be differences in fire frequency and seasonality of burning between land use areas that will have influenced woody cover at repeat photo sites. Results of fire, rainfall and land use patterns will be used to address the third objective to determine how these variables influence changes in woody cover.

3. To investigate how fire seasonality, frequency, land use and rainfall influence woody cover change and structure.

Changes in woody vegetation and structure (proportions of woody cover < 3 m and > 3 m) across land use areas are typically influenced by fire seasonality, frequency and rainfall (van Wilgen et al. 2014; Govender et al. 2006). However, this response is influenced by the species composition present at each of the photo sites (Shackleton et al. 1994). Therefore, outlying sites as determined in objective one's species composition cluster analysis were removed from analysis of response to rainfall and fire. I hypothesised that higher rainfall levels would lead to an increase in total woody cover as well as the cover of plants > 3 m in height (Staver et al. 2011; Scholes 2003; Sankaran et al. 2005). However, frequent late dry season fires have the opposite effect on vegetation in the > 3 m category when compared with effects of rainfall (Humphrey 2018). Frequent late dry season fires act as a limiting factor to vegetation height by encouraging re-sprouting and preventing the growth of mature trees (Humphrey 2018; Higgins et al. 2000). If late dry season fires and high fire frequency are a significant determinant of woody cover change then I hypothesised that photo sites with a relatively high level of late dry season fires and high fire frequency would display a majority cover of woody vegetation < 3 m and low levels of > 3 m. In contrast, early dry season fires reduce the extent of late dry season fires by maintaining a low fuel load availability (Humphrey 2018; Higgins et al. 2000). If areas have higher levels of early dry season fires, then there would have been a reduction in late dry season fires. Therefore, woody cover < 3 m in these areas would be lower and > 3 m woody cover would be higher. A low fire frequency typically allows woody cover to grow > 3 m by encouraging the development of mature trees and preventing re-sprouting (Higgins et al. 2000). If an area has a low fire frequency, I would expect that most of the woody cover will occur within the > 3 m category.

4. To explore the management and conservation implications of results from objectives 1-3.

In an attempt to synthesise results in relation to the specific goals of BNP the final objective will involve a discussion of the implications of this analysis for management and conservation. Specific mention will be made of what goals are affected by results of analysis and considering their implications, what interventions would be advisable by BNP.

CHAPTER 2: STUDY SITE AND METHODS

2.1. Study Site

Bwabwata National Park (18.1157° S, 21.6696° E; 627, 412 ha; 200 km wide, 32 km long) is situated at the core of the Kavango/Upper Zambezi Transboundary Conservation Area (KAZA-TFCA) in north-east Namibia (Figure 2). BNP, formerly known as the West Caprivi Nature Park, was established in 1963 and was officially re-proclaimed in 2007 (MET 2013). The park is bordered in the west and east by perennial rivers, the Okavango and Kwando Rivers respectively, which have their origins in the Angolan Highlands. The north and south boundary of BNP is created by the international borders of Angola and Botswana. South of Botswana's border lies the Okavango Delta, a World Heritage Site since 2014 (Scudder 2008) and to the North, Angola, the Luiana Partial Nature Reserve.

Before re-proclamation of BNP, the area was known as the Caprivi Strip or Western Caprivi, a name formed during South African administrative rule in Namibia (1915 - 1989). The name 'Caprivi' originated from the German Chancellor of the late 19th Century, Count Leo von Caprivi who, during the Berlin Conference of 1884-85, managed to obtain the strip in order to gain access to the Zambezi (Fisch. 1999). Recently, there has been increasing recognition of the need to remove names whose history represents colonial occupation and hence, the name Caprivi in reference to this area is considered inappropriate by many Namibians (Taylor 2012). Today the park is named after a village on its northern boundary, Bwabwata, meaning, 'the sound of bubbling water' in Khwé-dàm (Khwe language of the Khwe people resident in the MUA of the park).

2.1.1. Climate and Fire

BNP is situated in a semi tropical climate (west ≤ 550 mm – east > 650 mm). A rainfall gradient exists from west to east with average annual rainfall (calculated from '96/'97- '18/'19) from Namibia's meteorological station (http://www.meteona.com) in Rundu (180 km west of BNP) of 554 mm and Katima Mulilo (110 km east of BNP) of 610 mm. In north-east of Namibia, BNP contains the highest rainfall in the country with a uni-modal rainfall season that occurs from October to April (Figure 1). Temperatures reach seasonal maximum at the end of the dry season (October) and seasonal minimum during mid-winter (June-July) (http://www.meteona.com). The temperature and rainfall seasons influence the timing of what is considered early dry season (EDS: April -July) and late dry season burning (LDS: August - October) in BNP (Mendelsohn & Roberts 1997; Humphrey 2018; Govender et al. 2006).

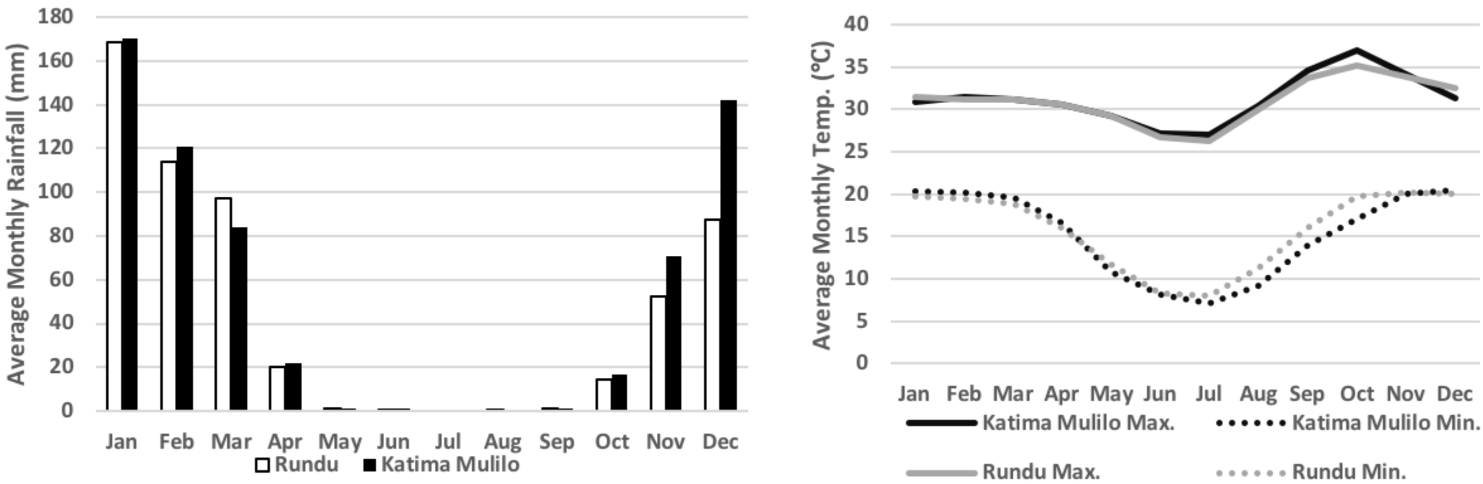


Figure 1 *Rainfall (left) and temperature (right) seasonality of Bwabwata National Park derived from the Namibian Meteorological Stations in Rundu and Katima Mulilo (http://www.meteona.com) for the years 1990 – 2019. The end of the dry season (October) is the hottest time of year when moisture availability is at its lowest, creating good conditions for fire to spread. Due to the uni-modal rainfall season, subsequent rainfall years are calculated from the beginning of the rainy season (October) to the end of the dry season (September).*

2.1.2. Geology, Landscape Features and Vegetation

BNP lies within the Kalahari basin of mid-Mesozoic (250–65 MYA) origin, which forms a large mostly sandy inland depression (Mendelsohn 1997 & 2009). Over time wind has shaped the sand into linear orientated dunes, which are today stabilised by vegetation (Mendelsohn 2009). In the depressions between linear dunes, heavier clay soils have accumulated, creating what is today a landscape of SE-NW oriented dune crests and low-lying intervening depressions known as Omurambas (Figure 2). The geomorphological history of these landscape features has created different environments resulting in heterogeneous vegetation patterns. Omurambas, with nutrient-rich sandy clay-loam soils and a higher water table, are characterised by Omuramba Grassland as defined by Mendelsohn and Roberts (1997). Omurambas hold the most promise for agricultural practice due to their nutrient and water availability and thus are selectively transformed into fields near settlements in the park (Mendelsohn & Roberts 1997). On the crest of the dunes, the sediment is characterised by a lower nutrient sandy composition which facilitates higher levels of drainage than the lower-lying Omurambas. The lower water table favours the root system of woodland tree species and thus the dune crests are characterised by woody cover. The mosaic between grassland in the Omurambas and woodland on dune crests forms the broader vegetation unit of BNP known as Kalahari woodland (Mendelsohn & Roberts 1997). Woody species which typically occur across BNP are *Burkea africana*, *Combretum imberbe*, *Pterocarpus angolensis*, *Schinziophyton rautanenii*, *Terminalia sericea*, *Acacia* *erioloba*, *Baikea plurijuga*, *Guibourtia coleosperma*, *Baphia massaiensis* and *Bauhinia urbaniana* (Mendelsohn & Roberts 1997; Full species list and authorities in Appendix 1C).

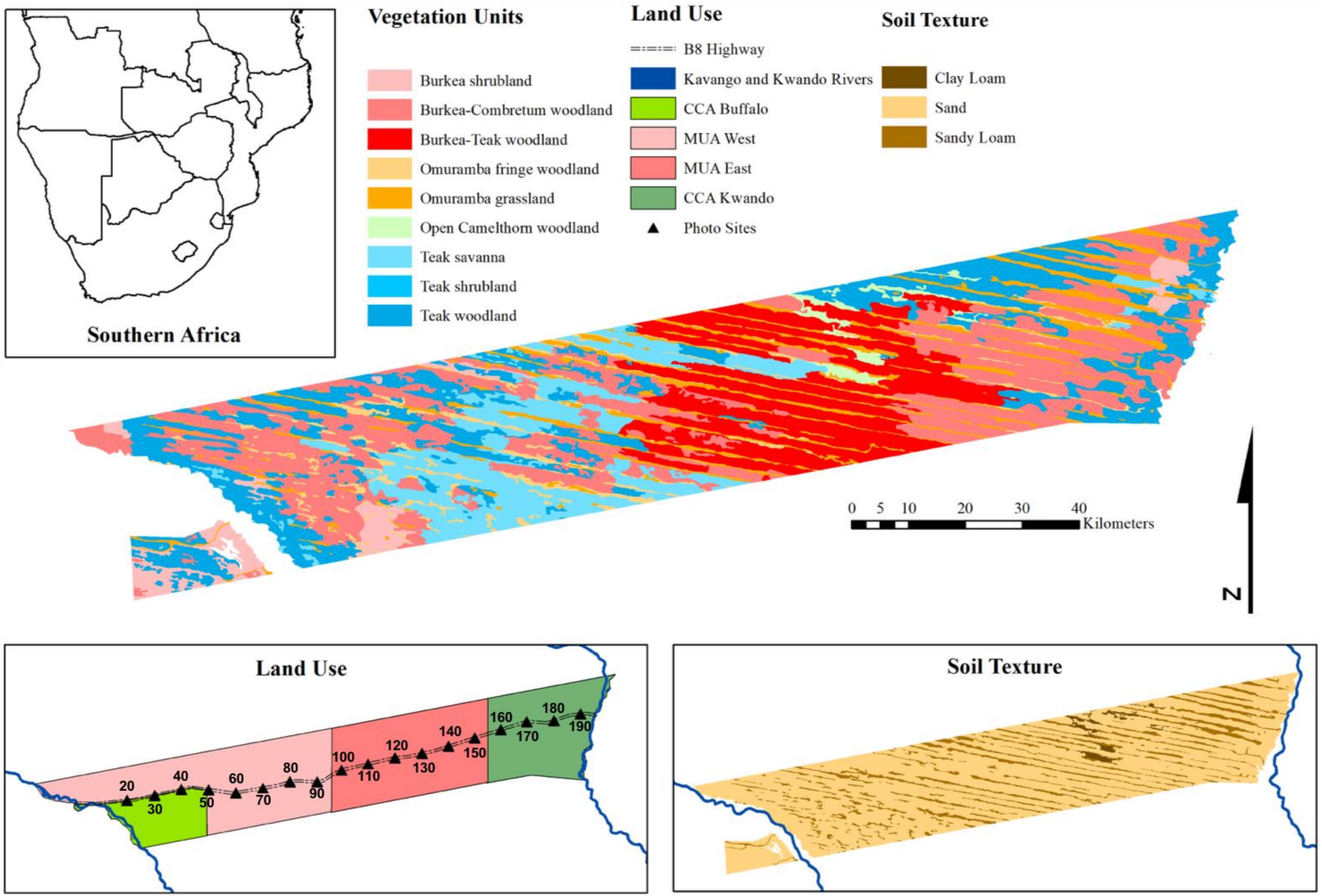
2.1.3. Land Use Categories:

The government agency with mandate over BNP is the Ministry of Environment and Tourism (MET) (Hitchcock & Vinding 2004). MET refers to BNP as the 'People's Park' due to the high amount of people and wildlife in the area (Paksi & Pyhälä 2018). An expression of interest to adopt a Community Based Natural Resource Management (CBNRM) approach to BNP has been made by the MET, however, full implementation of such a management approach is yet to be realised (Paksi & Pyhälä 2018; MET 2013). The park has been split into two primary management categories: Core Conservation Areas (CCA; 2219 km^2) and Multiple Use Areas (MUA; 4055 km^2) (Figure 2).

The CCAs are traditional conservation areas where no settlement is allowed, hunting and gathering is prohibited and the primary concern for MET management is the preservation of biodiversity and wildlife habitat (MET. 2013). In the MUAs, limited human settlement and development are allowed where community-based tourism, trophy hunting, foraging and agro-pastoralism activities are undertaken to sustain local community livelihoods (MET 2013). There are two land-use types in BNP, Multiple Use Areas (MUAs), where people reside, and Core Conservation Areas (CCAs), which are dedicated to biodiversity conservation. There are two MUAs, East and West, which lie in the centre of the park. The Conservation Areas comprise Buffalo in the west and Kwando in the East. (Figure 2). The MUA East and West are split at the 90 km mark in the park. The two CCAs are established along the river systems: Buffalo is situated east of the Okavango Riverand the Kwando CCA in the east of the park along the Kwando River (Figure 2) The MUA is situated in the middle of BNP and straddles the political constituent borders of the Kavango East and Zambezi regions (Immanuel. 2014).

As of 2009, the population living within the park was approximately 5 500 people, although the current population is probably higher as there has been an increased migration of certain groups moving into the MUA (Massyn et al. 2009; Suzman 2001). While up to date census' are not forthcoming, the split of the population between Khwe (former hunter-gatherers) and Mbukushu (agro-pastoralists) is approximately 82% and 18%, respectively (Dain-Owens 2010; Humphrey 2018).

Legend:



Southern Africa
Vegetation Units
Burkea shrubland
Burkea-Combretum woodland
Burkea-Teak woodland
Omuramba fringe woodland
Omuramba grassland
Open Camelthorn woodland
Teak savanna
Teak shrubland
Teak woodland
Land Use
B8 Highway
Kavango and Kwando Rivers
CCA Buffalo
MUA West
MUA East
CCA Kwando
Photo Sites
Soil Texture
Clay Loam
Sand
Sandy Loam
0 5 10 20 30 40
Kilometers
N
Land Use
20 30 40 50 60 70 80 90 100 110 120 130 140 150 160 170 180 190
Soil Texture

***Figure 2:** a) Continental perspective of the location of Bwabwata National Park in southern Africa, b) vegetation units. c) The fixed-point repeat photography sites are distributed within the inland section of the park between the Okavango and Kwando River systems and land use units. d) Soil texture map (correspondence between Omuramba grassland and sandy loam soil type and the occurrence of woodland on sandy soil is evident) (data derived from Mendelsohn and Roberts. 1997).*

The Bantu speaking Mbukushu's history and land use practice is closely tied to the Okavango River where agro-pastoralism holds more promise than the sandier and drier inland section of the park. Hence the vast majority of the Mbukushu population is considered to reside along the river systems or in the western section of the park, and the Khwe in the east (Humphrey 2018). The Khwe's dominant land use practice historically has been mobile hunting and gathering away from the river systems (Taylor 2009). Despite the traditionally mobile nature of living, a portion of the Khwe population has settled predominantly around Chetto village in the east of the park, where agro-pastoralism is practiced by both the Khwe and Mbukushu (Humphrey. 2018; Paksi & Pyhälä 2018). Thus, there is overlap in land use practices, and in addition there is to some extent mixed settlements of Mbukushu and Khwe in BNP. However, the MUA West and East can be considered to represent different fire management methods with the Khwe (MUA East) using fire to protect wild food products and the Mbukushu (MUA West) using fire for clearing of fields to cultivate crops (Humphrey 2018).

Despite being the minority population having a shorter history of settlement in the area (since the late 1800s), the Government of the Republic of Namibia (GRN) granted the Mbukushu a Traditional Authority (TA) status, and denied the Khwe a TA (Dain Owens et al. 2010). This power inequality can be considered as a continuing trend of Khwe subordination to the Mbukushu as there has historically been tension between the two groups (Taylor 2009). The primary motivation in the GRN's lack of recognising Khwe's TA is as a result of the recruitment of the Khwe by the South African National Defence Force (SANDF) during the late 20th century Namibian Liberation struggle (Taylor 2012). Thus, the GRN's *Traditional Authorities Act 17 of 1995* and *Traditional Authorities Act 25 of 2000* considers the Mbukushu elected Chief Mbambo as the representative of the Khwe when in fact the Khwe have their own Chief (Humphrey 2018 & Taylor 2012). This lack of Khwe sovereignty recognition has allowed for continued migration of Mbukushu agro-pastoralist practice further east into historical Khwe hunting and gathering territory (Humphrey 2018; Taylor 2009 & 2012). There is currently no census data to quantify the Mbukushu migration and increase in agro-pastoralism further east. This political tension and Khwe marginalisation will likely increase agro-pastoral practice in the MUAs. Despite the BNP management plan stating that there should be a full restriction of cattle movement in the park, cattle livestock can be seen walking freely in the MUAs of Mbukushu settlement (MET 2013). This trend is of concern due to the negative impacts associated with increasing cattle population and agricultural practice in conservation areas (MET 2013).

Cattle livestock can have multiple negative effects on conservation practice, however, a primary concern of this study is the increased grazing pressure which cattle create as well as their potential to promote the spread of selected woody species (Woinarksi et al. 2001; Brown & Archer 1987). The social dynamics in BNP have proven to be an interesting case study displaying how political history plays a role in influencing BNP's objective of biodiversity conservation (Humphrey 2018; MET 2013). Thus, monitoring potential impacts of differing land use practises and increasing Mbukushu migration into the centre of the park should be prioritised in the future within BNP.

2.2. Methods

Objective 1: Investigating the nature, extent and rate of change in woody cover (1999 – 2019) using repeat photography and remote sensing products.

In November 1999, 38 repeat photo sites were established by Garth Owen-Smith and Lynn Halstead of Integrated Rural Development and Nature Conservation (IRDNC), as part of a fire and vegetation monitoring project. The photo sites were located along the B8 highway running through BNP (Figure 2). These photo monitoring sites offered the opportunity to monitor the extent of woody cover change over time. From the western border in Buffalo CCA, through to the eastern most CCA, Kwando, there are two photo sites north and south of the B8 at 10 km intervals (see figure 3). Each site was marked by a steel rod and consists of photos taken in the four cardinal directions (N, E, S and W). The sites are located approximately 200 m north and south either side of the road, and therefore the road is not visible in the images. Thus any possible influence the road might have had on woody cover change can be considered negligible (Smit & Asner. 2012). Smit & Asner (2012), tested the influence of roads on woody cover elsewhere in savanna and found that there was a noticeably (2% – 6%) higher woody cover 0 – 15m away from the road compared to areas over 15m away from the road.

Thirty seven of the sites were revisited in late September and early October 2019 to recapture the photos taken in 1999. The photos of 1999 were repeated by relocating Global Positioning System (GPS) points of each site, which were also marked by steel rods placed in the ground. At each of the 37 sites revisited, a new GPS point was marked. The camera was placed on a tripod at a height of 1.54 m. A Canon 5D Mark II single lens reflex camera was used with the capacity for full frame at a 21 megapixel resolution. All photos were taken at 24 mm and then again at 40 mm using a Canon 24 –

105 mm image stabilised lens. Camera aperture and ISO was kept consistent at 8.0 and 100, allowing shutter speeds to vary according to light conditions. The camera and settings used in 1999 were not recorded. Notes were taken at each site, which included: weather conditions, time of photograph, general description of the location, photographic/camera settings, and time taken for each site's data collection (see Appendix 1C for an example). Each photo was stored in RAW (unprocessed by camera) and JPEG format at maximum resolution.

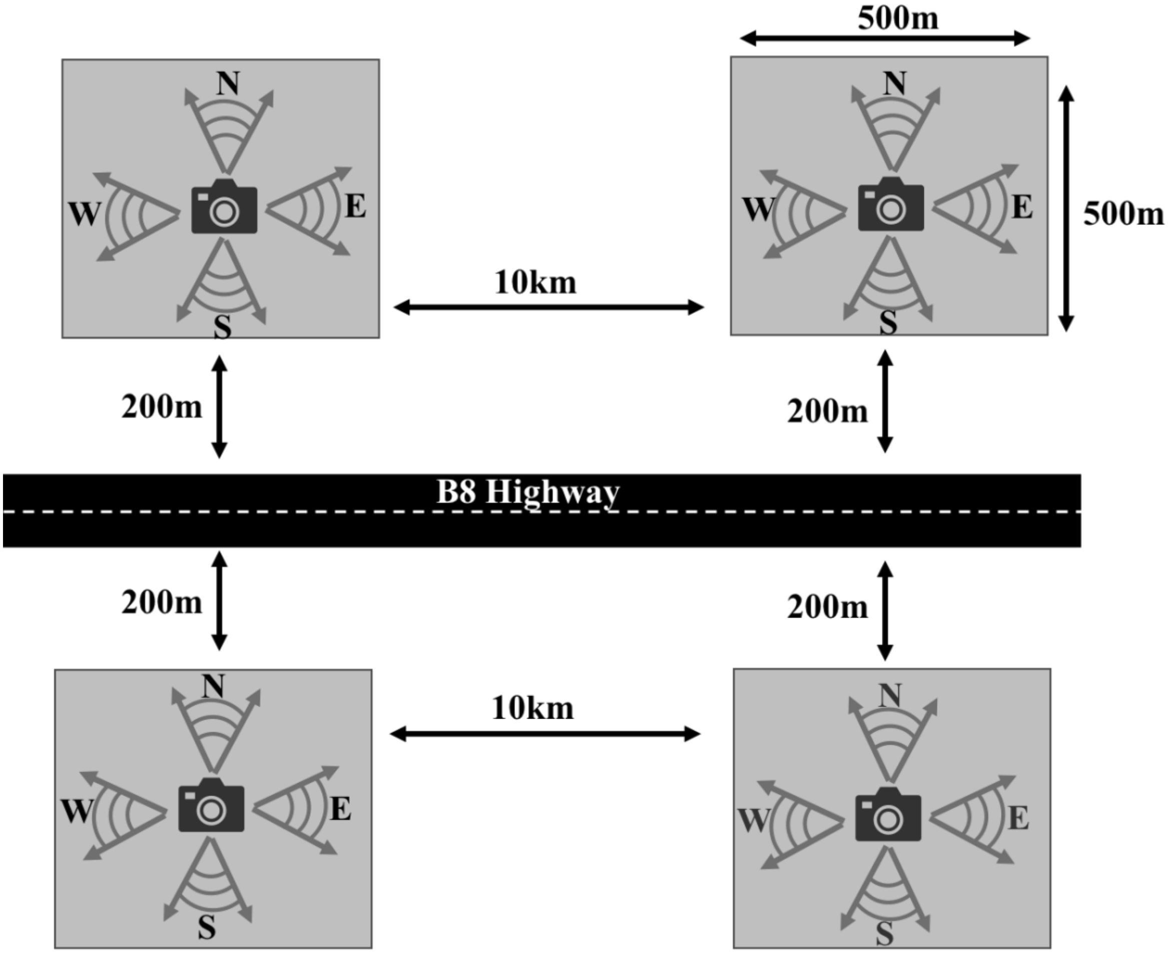


Figure 3 *Reduced scale depiction of the photo and EVI site design. The B8 Highway is 190kms long and there are in total 38 photo sites in pairs along the highway every 10km. At each site depicted by a camera, a photo was matched and repeated in all four cardinal directions. Sites were on average, 200m away from the road. Square outlines indicate the 500m block drawn around each photo site on GEE which EVI values were derived from.*

2.2.1.1 Field Vegetation Survey

A major strength of repeat photography studies is the ability to understand species-specific information (Webb et al., 2010). At each site the field researchers walked 'through' the photograph in each cardinal direction to determine the woody species composition and evaluate their structural growth forms (e.g. > 3 m and < 3 m). The distance that was walked through and considered for each vegetation survey at each fixed-point site varied from photo to photo due to the varying densities of woody cover across the park. The maximum distance walked in each cardinal direction was 200 m. This maximum was reached based on how far away woody species individuals could be identified from through the camera viewfinder. A research assistant placed a 3 m high ranging rod 10 m away from the camera. The ranging rod was used as a measure for the photo and field based estimates of woody cover of > 3 m and < 3 m. Estimates of cover were made using a comparison chart for reference as advised in Anderson (1983) (see Appendix 1A). The chart aids visualisation of vegetation cover into categories of 1%, 5%, 10%, 20%, 40%, 60% and 80%. While estimating cover, the researcher would refer to the chart to decide where the vegetation cover in question falls within these categories. Following walking through each photo, two researchers stood at the position of the camera while another stood in the photo frame. The researcher stationed within the photograph ensured that cover estimates were attributed to the correct woody species, particularly when a copse (i.e. a small stand of shrubs and/or trees) of multiple species was being evaluated. In order to reduce subjectivity, the two researchers stationed at the camera reached consensus on the relative cover estimates of: species present, woody cover > 3 m, and < 3 m and total woody cover. Species cover estimates and composition data were also used to develop abundance data for each site, which was used in a cluster analysis. This species composition list was also compiled to determine which woody plant species are responding to changing fire and land use dynamics and contributing to woody cover change in BNP.

The approach of using an infield visual cover-based method (Anderson 1983) is the most efficient way to collect woody cover abundance data over a large area such as the study site. The results should be considered as a relative estimate of species cover. A quantitative transect method such as point based intercept is considered more replicable (Godinez-Alvarez et al. 2009), however, such methods are not always more objective and decrease the sample size achievable in a given time (Dethier et al. 1993). Dethier et al. (1993) found that random point intercept methods either miss rare species or if the line intercepted them, overestimate rare species abundance. Furthermore, the photo survey performed in 1999 did not include any vegetation surveys so detailed information from 2019 would not be able to contribute towards the primary objective of understanding change in woody cover. Thus, the value of collecting a larger sample size and repeating detailed accounts (average time spent per photo site was 106 minutes) of all the photos from 1999 using visual estimates were considered favourable for the purposes of this study. The area of each photo site was less than 0.16 km^2. The total area surveyed covered 5.92km^2, which represented <1% of BNP's total area.

At each site, a general ecological description was recorded (see Appendix 1C). This involved establishing the vegetation type as defined by Mendelsohn & Roberts (1997), recording altitude, noting evidence of herbivory and fire and using a printed 1999 image of the site to document what major changes were evident. These field observations were not utilised in any statistical analysis of the project, but rather to provide empirical references to woody cover change and to corroborate trends found in the subsequent statistical analysis of land use, rainfall and fire dynamics in relation to woody cover change in BNP.

1.1. Species Abundance and Cluster Analysis: Grouping Sites

In order to explore woody species abundance in relation to fire and rainfall dynamics a cluster analysis was performed. The cluster analysis was based on the in-field vegetation survey of woody species abundance performed at each photo site. A species' response to disturbance or resource availability will differ according to characteristics defined by its taxonomy (Shackleton et al. 1994). Therefore, the aim of the cluster analysis was to investigate species abundance in the four land use areas. The results were used to remove sites whose species abundance were considered as outliers so as not to confound analysis of the species response to the disturbance (FRI, EDSBA, LDSBA, Land Use) and rainfall as an environmental variable (MAP) tested for objective 3.

The vegan package (Oksanen et al. 2019) was used to perform a hierarchical cluster analysis based on the Bray-Curtis similarity measure with data of species abundance per site. Each cardinal directions' species cover scores derived during the field vegetation survey were averaged to create a single mean species abundance measure for each site. The Bray – Curtis measure was selected as it is considered an appropriate method to detect underlying ecological gradients (Gardner 2014). Once cluster groups were defined, relative species cover within each group was calculated in order to determine the percentage abundance at which species occurred within each grouping.

1.2. Woody Cover Photo Estimates

Woody cover was separated into three categories: total woody cover, < 3 m woody cover and > 3 m woody cover. The field-based cover estimates performed in 2019 were used to calibrate estimates of woody cover in 1999 (see Appendix 1A: photo scoring scale sheet for < 3 m and < 3 m [2019; 1999]) as the series of photos taken in 1999 were not accompanied by any field-based observations.

The woody cover scoring process was performed after the field data collection by a panel of three ecologists to minimise subjectivity. Two large screens were used to display the photos from 1999 and 2019 for comparison purposes. Each ecologist then scored the photo independently without being influenced by the rest of the panel. Once each ecologist had produced their independent score, it was then compared to the other two ecologists' scores. If there was a difference of scores, the ecologist's perspective behind the different cover score was discussed and a consensus woody cover value was reached by the panel. Independent woody cover scores were made in five percent increments to reduce false precision. However, upon a difference in independent scores (> 5%) within the panel and following a discussion on these differences, a mid-way point was sometimes necessary for photos which did not fit the scale sheet perfectly and hence, consensus values sometimes had a finer scale than 5% increments. This process was performed first for the scoring of total woody cover to allow for consensus on the total cover value before separating this total value into woody cover < 3 m and woody cover > 3 m.

Each photo was compared to a scale sheet (Appendix 1A) which comprised a visual scale and a photo scale. The visual scale is an adaptation of Anderson's (1983) foliage cover chart which the research team used as a comparison for visual estimation. The photo scale was developed by using the infield estimates of woody cover in 2019 to score photos from 1999 with similar cover to aid initial scoring. Thus, infield estimates were the baseline upon which estimates of cover in 1999 were performed. This allowed for consistency in scoring and prevented subjectivity or scorer fatigue from affecting the visual estimation as once a full range of possible cover scores were derived for the scaling sheet, scoring photos was a matter of comparing the current photo to the scale and deciding which photo from the scaling sheet was the most similar. While photos used in the scale were selected for their clarity of vision, a shortcoming was that any woody cover in the foreground of photos obstructed the view of vegetation behind them. In this case, field notes were referred to and special attention was paid to possible discrepancies between in field estimates and photo estimates that may have arisen as a result.

A count of sites with > 3 m woody cover (canopy cover/area) equal to or higher than 40% and 50% was performed for 1999 and 2019. Thus, results of woody cover estimates from the study are placed within context of the woody vs grassy state savanna threshold as described by Staver et al. (2011) and Scholes (2003). The > 3 m woody cover category was considered to represent tree cover and allowed comparison with the thresholds of 40% and 50% (Staver et al. 2011; Scholes 2003). While a question of woody state savanna threshold is best answered by the results of the > 3 m woody cover, a count was also performed for categories of < 3 m woody cover and total woody cover. The split of woody cover to < 3 m and > 3 m allowed a coarse understanding of height structure which was used to distinguish between shrub encroachment and changes in tree cover (> 3 m). Therefore, these categories are used to evaluate the effect of fire frequency and seasonality on the cover and structure of woody species as well as the total woody cover (sum of < 3 m and > 3 m woody cover).

1.3. Data Analysis: Pairwise T-Test for Woody Cover Change (1999 – 2019)

A pairwise t-test was performed on photo cover scores for all sites across BNP from 1999 and 2019 to establish if woody cover differed over the 20 years (1999 – 2019) of interest. During fieldwork, sites 70N, 110N and 110S were not able to be rephotographed due to community settlement having converted the site to an agricultural field or housing being built over the site. The total number of sites was, therefore, reduced from 36 to 33.

This test was performed separately for total woody cover, woody cover < 3 m and woody cover > 3 m. All statistical analysis was performed in the statistical software package of R (R Development Core Team 2017). With the null hypothesis being no woody cover change between 1999 and 2019, a significant ($P < 0.001$) difference of the means indicated a change in woody cover with a ninety nine percent confidence interval (CI). Photo cover scores from each cardinal direction at a site were averaged in order to obtain one mean value of woody cover for each site at each time interval. Each site was considered as independent and thus sample size for the test of woody cover change over the whole BNP was 37.

In order to understand if there were differences in woody cover change across the land use categories a pairwise t-test was also performed for each land use category of: Buffalo CCA, West MUA, East MUA and Kwando CCA.

1.4. Enhanced Vegetation Index (EVI) and Google Earth Engine (GEE):

A weakness of repeat photographic methods is the lack of quantitative data that can be determined from photos (Webb et al. 2010). Thus, in order to test the results of woody cover change based on the repeat photo woody cover estimates, bi-weekly EVI values for photo dates in 1999 and 2019 were extracted using Google Earth Engine (GEE). This was performed to explore whether there was a correlation between vegetation productivity and the photo cover estimates. The addition of EVI based measurement allowed results of woody cover change derived from a qualitative (in field and photo based woody cover estimates based on Anderson (1983)) method to be tested against that of a quantitative method (EVI derived from GEE). While it is acknowledged that the use of trained remote sensing product that excludes grass in measurement would have been beneficial, time restraints of the research meant that such training and product use were not realistic. However, the lack of remote sensing measurement would have reduced the reliability of the results and therefore EVI was used. While EVI is a measure of productivity which includes grass, both measurements used (1999 and 2019) were taken during the end of the dry season when grass cover is at its lowest and most woody species are able to retain some level of green.

EVI scores for a surface area of 500 m around each site (Figure 3), were derived from the Landsat 8 and 4 collection 1 product, supported by the National Aeronautics and Space Administration (NASA) (Chander, Markham & Helder 2009). Landsat satellites extract spectral information received by radiation reflected from vegetation on the earth's surface and this information is transformed into retrievable data in spectral bands (e.g. near infrared, infrared and blue) (Huete et al. 2002; Moore et al. 2016). The EVI product returns information about vegetation properties (e.g. photosynthetic activity, tree canopy structural variations), and therefore provides a measure of temporal and spatial variation in vegetation phenology. These measures provide insight into productivity and most importantly for this study, a measure of cover (Huete et al. 2002). Similar to the Normalised Difference Vegetation Index (NDVI) also derived from Landsat, EVI allows for the monitoring of photosynthetic activity at a resolution of 500 m with a 16-day compositing period (Huete et al. 2002). EVI was developed after NDVI in order to be able to improve ability to differentiate between canopy background signal and the canopy itself as well as to be less sensitive to detection interference caused by atmospheric activity (Huete et al. 2002). When compared to the NDVI product, EVI detects less atmospheric noise and is considered an improved product (Huete et al. 2002) The improved ability of EVI to account for atmospheric disturbance is relevant to application of EVI in the BNP due to the large amounts of smoke from fire and large cloud formations during the rainy season which would likely reduce confidence in index values (Huete et al. 2002). EVI was calculated by the following equation (Huete *et al.* 2002):

$$\mathrm{EVI} = G \frac{\rho_{\mathrm{NIR}} - \rho_{\mathrm{red}}}{\rho_{\mathrm{NIR}} + C_1 \times \rho_{\mathrm{red}} - C_2 \times \rho_{\mathrm{blue}} + L}$$

Here, ρ is surface reflectance of near infrared(NIR), blue and infrared(red) radiation that have been partially or fully corrected, L is the adjustment used to correct for canopy background (L=1) , G is the gain factor (G=2.5) and C_1 and C_2 are aerosol resistance coefficients (C_1=6;C_2=7.5). This equation returns an index value which is essentially a measure of vegetation greenness between -1 (least green) and 1 (most green) (Huete et al. 2002; Moore et al. 2016). All Landsat data processing was completed on the web-based remote sensing platform and cloud computing technology of GEE. GEE is online and open access performing as a Geographical Information Systems (GIS) platform facilitating access and analysis of geo-spatial data (Google Earth Engine Team 2015; https://earthengine.google.org/).

Change in bi-weekly EVI values from the date of photos taken in November 1999 and October 2019 was plotted against change in woody cover photo estimates in order to understand whether the two measures of cover change correlated. A Pearson correlation co-efficient was used as a correlation test using R (R Development Core Team 2008). The ggpubr package (Kassambara 2019) was used to plot the correlation. The test was performed for all 37 sites where a combination of the repeat photos from 2019 and 1999 were paired successfully.

Objective 2: Describing inter-decadal scale patterns in rainfall, fire, and land use at repeat photography sites.

2.1. Rainfall: CHIRPS Mean Annual Precipitation (MAP)

Climate Hazards Group InfraRed Precipitation with Station data (CHIRPS) is a global rainfall dataset covering 50^{0} S - 50^{0} N across all longitudes, with a temporal scale starting from 1981 to present. By interpolating weather station data with gridded satellite-based precipitation estimates, CHIRPS provide rainfall estimates at a resolution of 0.05^{0} by 0.05^{0} (Funk et al. 2015). For the purposes of this study, these fine resolution rainfall values were extracted for each pair of sites which were 10 km apart. Independence of the CHIRPS rainfall values allows for a fine scale understanding of the rainfall gradient which occurs from west to east of the park.

Using GEE, total monthly precipitation (mm) from 1996 – 2019 was extracted by drawing a 1.5 km by 1.5 km (1.5 km^2) size polygon over each pair of sites. Mean Annual Precipitation (MAP) was calculated for each pair of sites for the time period 1996 – 2019. Due to the uni-modal summer rainfall season, years were calculated from the beginning of the rainy season in October to the end of the dry season in the following year September.

In addition, MAP for the entire surface area of the park was calculated. This broader spatial scale rainfall value was considered in order to be able to place into context the antecedent rainfall of each photo year (Photo years: 1999 and 2019). Depending on the results of woody cover change analysis, higher antecedent rainfall will have had a positive effect on woody cover growth and vice versa for lower antecedent rainfall (Scanlon et al. 2002; Sankaran et al. 2005). Therefore, using the polygon feature of GEE monthly rainfall values for BNP were extracted from CHIRPS for 1996 – 2019. The MAP for each year (October – September) of rainfall was then compared against the average MAP for the period of 1996 – 2019, to be able to visualise low and high rainfall years.

In order to test the accuracy and/or bias of the CHIRPS dataset, a Pearson correlation co-efficient test (R Development Core Team 2008) was performed for observed rainfall data from the Katima Mulilo (east) and Rundu (west) weather stations (www.meteona.com) and CHIRPS data for the geographic locations of each weather station. The ggpubr package (Kassambara 2019) was used to plot the correlation.

2.2. MODIS Burned Area: Fire Frequency and Seasonality

To analyse the frequency and seasonality of fire at each photo site, the MODIS Collection 6 MCD64A1 Burned Area product was used (Giglio et al. 2018). This product maps the spatial and temporal extent of burning biomass at a resolution of 500 m from the year 2000 – present. This spatial resolution allows independent frequency and seasonality to be determined for each repeat photo site. The product was downloaded from the University of Maryland's server (ftp://ba1.geog.umd.edu) where window 13 (southern Africa) Georeferenced Tagged Image File Format (GeoTIFF) files were extracted for each month from 2000 - present. Unfortunately, there was a prolonged sensor outage during May – July 2001, and thus there is a gap in fire data for this period. The accuracy of date of burn from the product is within 8 days, which for the purposes of distinguishing between early dry season and late dry season is considered adequate (Giglio et al. 2018).

In order to extract the relevant GeoTIFF files to the area of the repeat photo sites, ArcGIS 10.4 software was used (ESRI 2011). Sites north and south of the road were considered as independent from each other due to the manmade firebreak of the B8 road which bisects the park and pairs of sites. To begin the process of extracting site-specific fire information, two separate shapefiles were added which comprised north and south sections of BNP. Around each site a 500 m buffer was created. In order to cut this buffer so that it did not include any fire on the opposite side of the road, a mask was created on each site which used the road as a cutting feature. Calculation of fire statistics for each site was done year by year in ArcGIS. The months of each year were added into the map and using the north and south layers, GeoTIFF files were cut between north and south of BNP, representing annual fire activity. The cell statistics tool was used on the north and south layers of each year, where the maximum value of the class field (representing the julian day of burn) was extracted. This method assumes that an area/pixel which burnt once in a year, cannot burn again. This assumption was accepted considering the uni-modal rainfall season which would theoretically prevent enough regrowth for a subsequent fire to occur before the next rainy season. The tabulate area tool was used on both the north and south layer. This method returns a table with the number of pixels burnt per site per year and the date of burn. The fire return interval (FRI) and fire seasonality (i.e. early and late dry season fires) were calculated for each site from this dataset.

In order to determine the frequency of fires at each site, the fire return interval (FRI) was calculated. Thus, for the purposes of this study, FRI is considered as a metric of fire frequency in that it represents the average time (years) that a fire takes to return to a site. A shorter FRI represents a higher fire frequency and a longer FRI represents a lower fire frequency. Fire metrics were calculated for the years 2000 – 2018 due to 2019 being an incomplete year at the time of analysis. By calculating the total number of fires which occurred at each site from 2000 – 2018 and then dividing the total amount of fire by the number of years, an average annual fire frequency (AAFF) was found. Using AAFF, FRI was calculated using the following formula:

$$FRI = (1 - AAFF)*18$$

Annual burnt area was calculated by dividing the total number of pixels burnt per year by the total number of pixels per site. Sites varied slightly in their pixel size due to curves in the road which altered the size of the mask created in ArcGIS. Since burnt area is always calculated as a proportion of the total area (Archibald et al. 2010), burned area is reported as a percentage of the site. In addition, the portion of the site burnt in the early dry season and late dry season was calculated for each site. This seasonal split is important as it allows for differentiation between fires occurring when vegetation is moist (early dry season; April -July), which results in small patchy burns and dry (late dry season; August – October) (Humphrey 2018) season when fires are large and hot, and spread easily due to conducive climatic conditions. The seasonality of burn is closely associated with the intensity of fire as dry biomass combusts quicker and thus late dry season fires typically burn more biomass (Humphrey 2018). Intensity as a function of seasonality is also supported by increased wind speed that occurs during the late dry season in BNP which allows fires to spread further and burn longer (Humphrey 2018). Thus, the average early dry season burnt area (EDSBA) and late dry season burnt area (LDSBA) was calculated to understand the effect of seasonality.

2.3.Fire Management Between Land Use Areas

There are contrasting dynamics of fire application involved in each land use category as a result of contemporary management and settlement history (Humphrey 2018). Thus, for the purposes of this study, each land use category: CCA Buffalo, MUA West, MUA East and CCA Kwando will be treated as independent and the differences among each land use category were tested to address the objectives of this study. The result of this sub-grouping will allow insight into the possible effects that differing land use practices may have on woody cover change. A summary of the different land use practice and characteristics between categories are seen in Table 1.

Considering the differences in the use of fire between the east and the west (MET. 2016) and the four land use categories of BNP (Humphrey 2018), it is important to test and confirm how fire management differs between the categories. A one-way analysis of variance using the dplyr package (Wickham et al. 2018), was performed on three fire regime variables for the purposes of this study: FRI, LDSBA and EDSBA. These variables were grouped according to the four land use categories: Buffalo CCA, West MUA, East MUA and Kwando CCA. Normality of the fire variable datasets was tested visually and confirmed. Levene's test was performed using the car package (Fox & Weisburg 2019), to determine whether the homogeneity of variance assumption was true and was subsequently confirmed. A Tukey Honest Significant Difference (HSD) test was performed on the means of fire variables across the groups of land use categories to establish whether the fire variables were significantly different across the categories. Here the null hypothesis was no difference between the means of the different groups and the alternative at least one land use categories' mean was different to the others.

Table 1*: Summary of dominant characteristics relevant to the study, defining the land use categories of BNP (MET. 2013; MET. 2016).*

Traits	CCA Buffalo	MUA West	MUA East	CCA Kwando
Size (km2)	629	2446	1609	1345
Land use	Protected Area	Settlement area	Settlement area	Protected Area
Human Population Size	0	Not known - majority of total MUA population of approximately 5500 (MET. 2016)	Not known - minority of total MUA population of approximately 5500 (MET. 2016)	0
Dominant Activities (MET, 2013)	Biodiversity conservation and controlled tourism	Mostly agro-pastoral with some hunter gathering, small mammal hunting and floral gathering, community-based tourism and trophy hunting. Clearing fields for agricultural purposes. Limited use of woody species for construction.	Hunter gathering and agro-pastoral, small mammal hunting and floral gathering, community-based tourism and trophy hunting. Clearing fields for agricultural purposes. Limited use of woody species for construction.	Biodiversity conservation and controlled tourism
Fire management (MET 2016)	Inconsistent, uncontrolled, mostly late dry season fires.	Inconsistent, uncontrolled, late dry season fires are dominant.	Consistent, early dry season burning occurs.	Consistent, reduced uncontrolled fires, early dry season burning occurs.
Game activity (MET 2013)	High concentration along river system (with notable presence of elephants during fieldwork period) predominantly in the dry season.	Low concentration - wet season is highest activity period.	Low concentration - wet season is highest activity period.	High concentration along river system (with notable presence of elephants during fieldwork period) predominantly in the dry season.
Domestic Animals	NA	Cattle, goats, donkeys, chickens and dogs.	Cattle, goats, donkeys, chickens and dogs.	NA

Objective 3: Investigate how fire seasonality, frequency, land use and rainfall drive woody cover change and structure

A Generalised Linear Mixed Model (GLMM) was selected to investigate the drivers of woody cover change with the use of the lme4 package available in R (Bates et al. 2015). This statistical analysis method was considered appropriate due to the method of data collection, since a Generalised Linear Model (GLM) assumes independence between samples, a GLMM controls for potential lack of independence by testing for independence between samples (Zuur, Hilbe & Ieno 2013). This test was required as the repeat photo sites, although situated at 10 km intervals across BNP, were paired north and south of the road approximately 400 m in distance from each other, and therefore cannot be considered independent samples (Figure 3). Therefore, at each photo site the four cardinal directions (position) that comprised the total change per site were included as crossed random effects creating four random effects per site. The formula for the model fitted by maximum likelihood was:

$$\textit{Woody Cover Change} \sim \textit{FRI} + \textit{MAP} + \textit{EDSBA} + \textit{LDSBA} + \textit{Land Use} + (1 \mid \textit{Position})$$

The GLMM was run using woody cover change as the response variable for each category of woody cover (Total, <3m and >3m). The response being tested was a change in woody cover calculated by subtracting the cover of 1999 from 2019 derived from the photo scoring procedure. Woody cover change for all three response variable categories was transformed by adding 50 to each photo score in order to remove negative numbers. The continuous explanatory variables were Fire Return Interval (FRI), Early and Late Dry Season Burnt Area (EDSBA and LDSBA) and Mean Annual Precipitation (MAP). These variables were derived from the results of objective two. Explanatory variables used in analysis did not correlate (Appendix 1B). Land use was used as a categorical explanatory variable. The distribution of the response variables were visualized using the car package (Fox & Weisburg 2011). Total woody cover change and > 3 m woody cover change were fitted with normal distribution. The < 3 m woody cover category was transformed using the log link function for a normal distribution. Results of the model were plotted using the effects (Fox 2003) and nlme (Pinheiro et al. 2018) packages which allowed for the visualisation and inspection of the predicted effects of the explanatory variables.

CHAPTER 3: RESULTS

Objective 1: The Nature and Extent of Change in Woody Cover (1999 – 2019)

Total woody cover in BNP significantly increased by 26% relative to 1999 (Table 2: See Figure 4 for examples). The majority of this increase in cover occurred within the < 3 m category, which revealed a 45% relative change in woody cover. In BNP, woody cover > 3 m increased relative to 1999 by 12% over the last 20 years. Across all land use categories, there was a significant increase in total woody cover apart from CCA Buffalo (p = NS; df= 2), however this may have been due to a low sample size in this land use area (CCA Buffalo contained 3 plots). Across the entire BNP, woody cover in the < 3 m category increased at almost twice the rate of the > 3 m woody cover.

Trends in woody cover differed among land-use categories. There was a significant increase in > 3 m woody cover in both CCA Kwando and the MUA East, while no change was detected in the MUA West in this height category. However, the MUA West showed the largest significant change with an increase in < 3 m woody cover across all land use categories. The < 3 m woody cover in MUA West increased at 4 times the rate of MUA East and over five times the rate of CCA Kwando. With regard to determining if there has been a transition from grass to a woody savanna, results revealed the number of sites with > 3 m woody cover larger than or equal to 50 % has increased from 0 in 1999 to 2 in 2019 (Table 3). A similar trend was found at sites for > 3 m woody cover greater than or equal to 40 %, where in 1999 there were two sites and nine in 2019.

The overall trend of an increase in woody cover was also captured by the number of sites from 1999 to 2019 which crossed the 40 % and 50 % threshold for total woody cover and < 3 m woody cover categories. However, the crossing of these thresholds by < 3 m woody cover and total woody cover is not considered a transition to a woody state as the categories in this study cannot be considered as representative of tree cover (Staver et al. 2011; Scholes 2003).

As depicted in Figure 5, sites which are increasing in > 3 m woody cover are predominantly in the east of the park while sites increasing in < 3 m woody cover are predominantly in the west. This pattern is evident in horizontal movement along the x-axis of > 3 m woody cover by arrows in blue (CCA Kwando) and yellow (MUA East) which is in direct contrast to the vertical movement along the y-axis of < 3 m woody cover by arrows in red (MUA West) and green (CCA Buffalo). Sites crossing the 40% threshold are considered in transition towards a woody state and those crossing the 50% threshold as woody savanna state sites. The only arrows to cross these lines are sites from the east of the park, a result which corresponds to results of the t-test (Table 2).

Table 1: *Results of the pairwise t-test returning the significance and change estimated from repeat photos of 1999 and 2019. The extent of change was tested by performing t-tests within each land use category. Summary statistics returned are the mean ($\bar{x}$) and standard deviation ($\sigma_{\bar{x}}$) ,change in mean relative to 1999, difference of the means and 99% confidence intervals, degrees of freedom (df), t-statistics (t-stat) and p-value (NS – non-significant) evaluating the null hypothesis that there was no difference in woody cover between 1999 and 2019.*

Land Use	Woody Change Category	1999 $\bar{x}$ ($\sigma_{\bar{x}}$)	2019 $\bar{x}$ ($\sigma_{\bar{x}}$)	% Change	Mean of differences (99% CI)	df	t-stat	p-value
CCA Buffalo	**Total**	38(9)	40(13)	+6	+2 (-33 – 38)	2	0.65	NS
	<3m	14(6)	24(12)	+41	+10 (-48 – 68)	2	1.74	NS
	>3m	23(7)	16(2)	-46	-7 (-48 - 33)	2	-1.82	NS
MUA West	**Total**	36(7)	51 (14)	+31	+16 (7 – 24)	13	5.60	<0.001
	<3m	11(6)	26 (12)	+59	+16 (5– 26)	13	4.64	<0.001
	>3m	25(8)	25 (8)	0	0 (-8– 8)	13	0.01	NS
MUA East	**Total**	43 (5)	53 (8)	+19	+10 (5 – 15)	7	6.77	<0.001
	<3m	8(2)	12(2)	+33	+4 (1– 7)	7	4.40	<0.01
	>3m	34 (4)	40(6)	+15	+6 (2– 10)	7	4.74	<0.01
CCA Kwando	**Total**	43 (8)	58 (10)	+27	+16 (3 – 28)	7	4.46	<0.01
	<3m	14(4)	16(4)	+15	+3 (-1 – 6)	7	2.59	NS
	>3m	29(5)	42(8)	+31	+13 (2 – 24)	7	4.17	<0.01
BNP	**Total**	43 (7)	53(12)	+26	+13 (9 - 17)	35	8.71	<0.001
	<3m	11(5)	20(11)	+45	+9(4 – 13)	35	5.10	<0.001
	>3m	28 (12)	32 (8)	+12	+4 (0 - 8)	35	2.53	<0.05

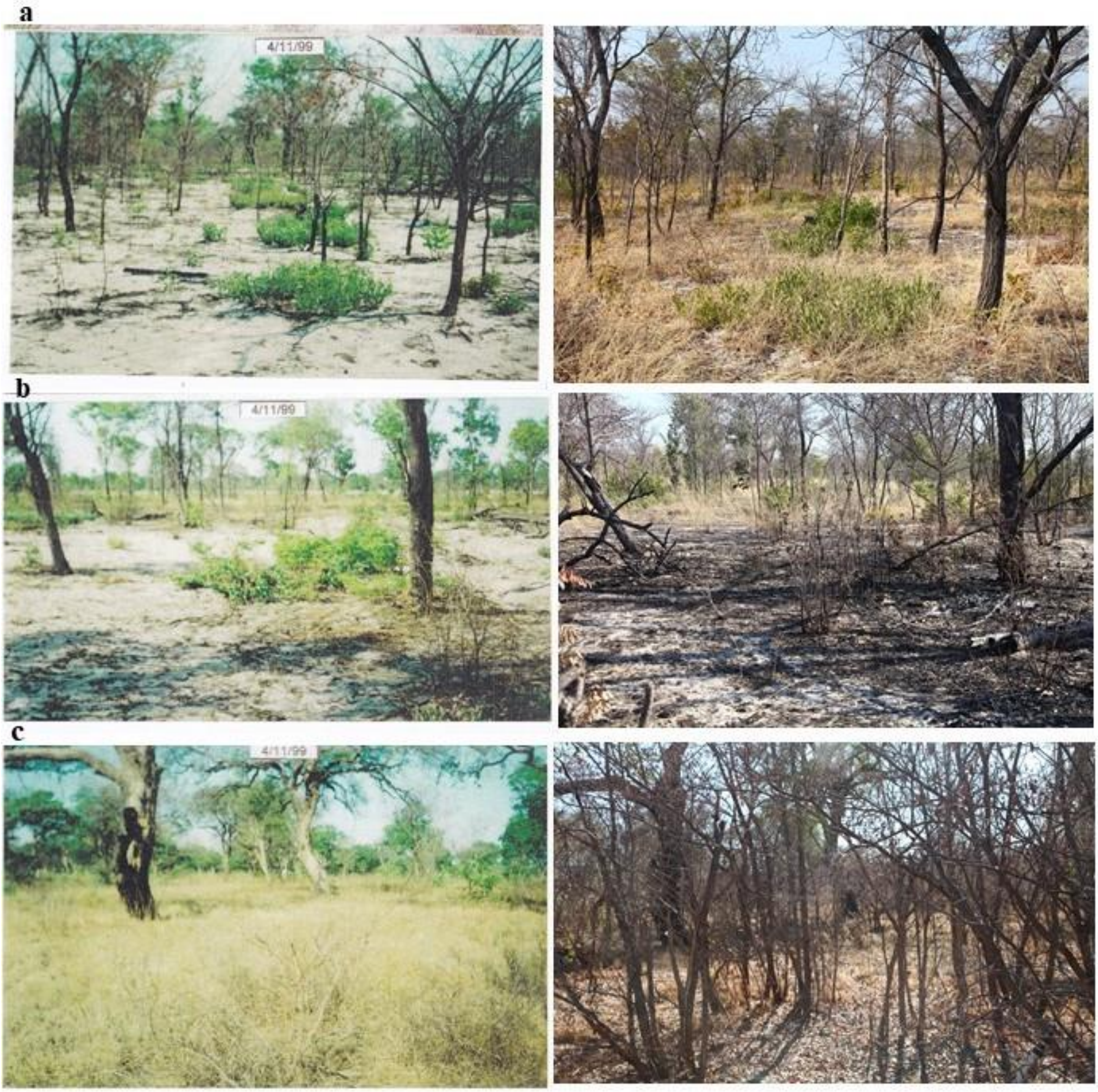


***Figure 4**: Examples of woody vegetation change seen at repeat photography sites on a scale from minimal change (a) to maximum change (c). Photos on the left are taken by Owen-Smith and Halstead in November 1999 and photos on the right are taken by the author in October 2019. Photo a represents a site where an increase in woody cover of 3% was determined by the panel of ecologists due to a marginal increase in Terminalia sericea individuals in the mid-ground. Photo b is an example of below average (10% increase in total woody cover) change with the whole of BNP seeing an average increase of 13% (Table 2). The majority of change in photo b is as a result of T. sericea thickening in the mid-ground. Photo c represents a maximum amount of change found at photo sites (36% increase in total woody cover). The two large Guibourtia coleosperma individuals are still present but are much more difficult to see as a result of an intense thickening of T. sericea which can be considered to have shaded out the swath of grass cover seen in the '99 image.*

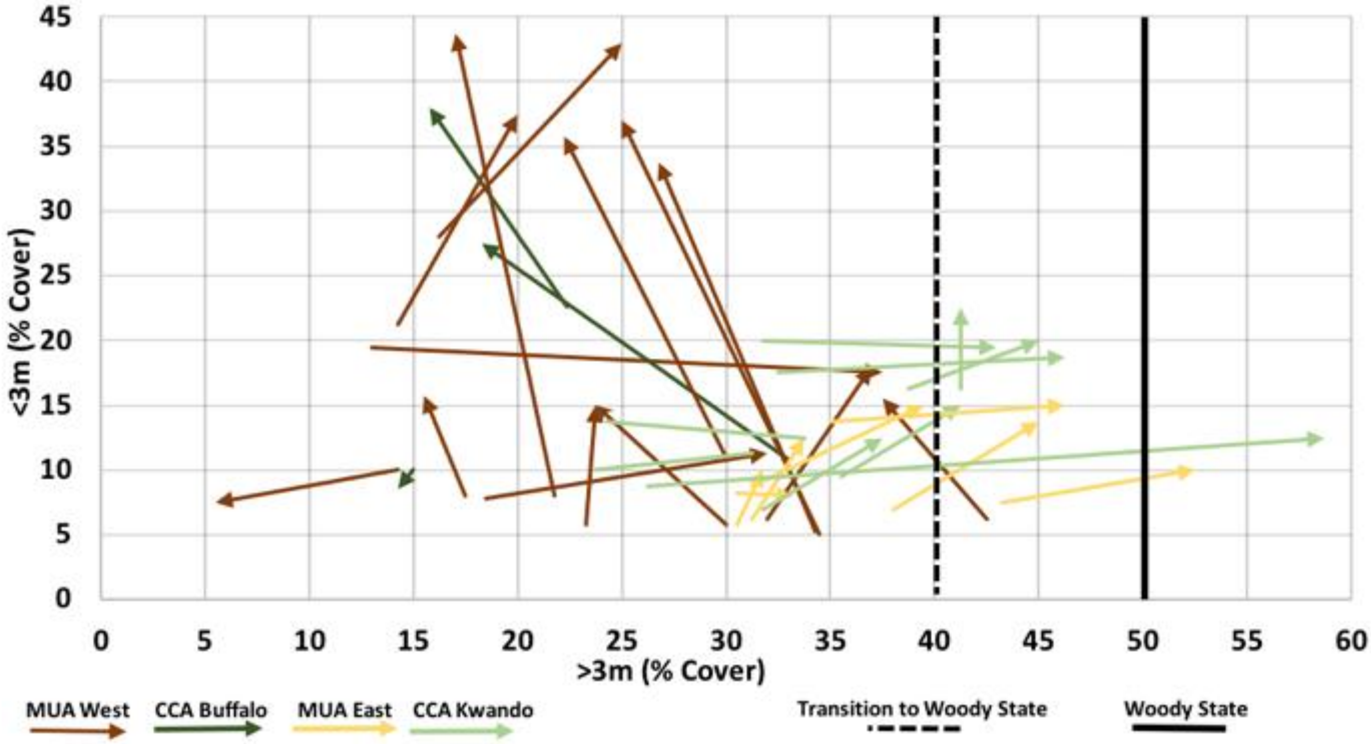


Figure 5: *Trajectory of change for each of the 37 sites where repeat photos were performed with arrow heads indicating the direction of change from 1999 to 2019. Each colour represents the respective land use category with lighter colours representing the eastern section and darker colours the western section of BNP. The grassy to woody savanna state thresholds are placed on the x-axis (> 3 m woody cover) which is considered as a proxy for tree cover.*

Table 2: *Count data of sites from 1999 and 2019 which have ≥ 40% and 50% cover for each category: Total woody cover, <3m woody cover and >3m woody cover. The >3m woody cover category is taken as a proxy for tree cover, which if above 50%, is considered to represent a site that has transitioned from a grassy savanna state to a woody savanna state (Staver et al. 2011; Scholes 2003).*

	1999	Woody Cover (%)		2019	Woody Cover (%)	
	Total	**< 3 m**	**>3m**	**Total**	**< 3 m**	**>3m**
Sites ≥ 50 % woody cover	4	0	0	22	0	2
Sites ≥ 40 % woody cover	14	0	2	31	2	9

1.1. Grouping Sites According to Woody Species Composition

Of the 37 sites where vegetation surveys were conducted, five were considered as outliers in terms of species abundance (Figure 6). These sites were removed from subsequent analyses in order to minimise the possibility that a species' unique response to disturbance would confound results of objective 3. In total sixty-four woody species were detected across all sites (Appendix 1C). Four species groups were identified with the cluster analysis. Group one and two contained the most number of photo sites and three and four, only five sites in total. This dendrogram provided the motivation behind selecting sites in group three and four as outliers when testing for woody cover as a response variable in the GLMM of objective 3. The spacing of these cluster groups across BNP can be seen in Figure 7 where with some exceptions, cluster group two seems to dominate in the west and cluster group one in the east. This spacing is helpful to visualise with the list of species which make up the respective cluster groups as is seen in Table 5.

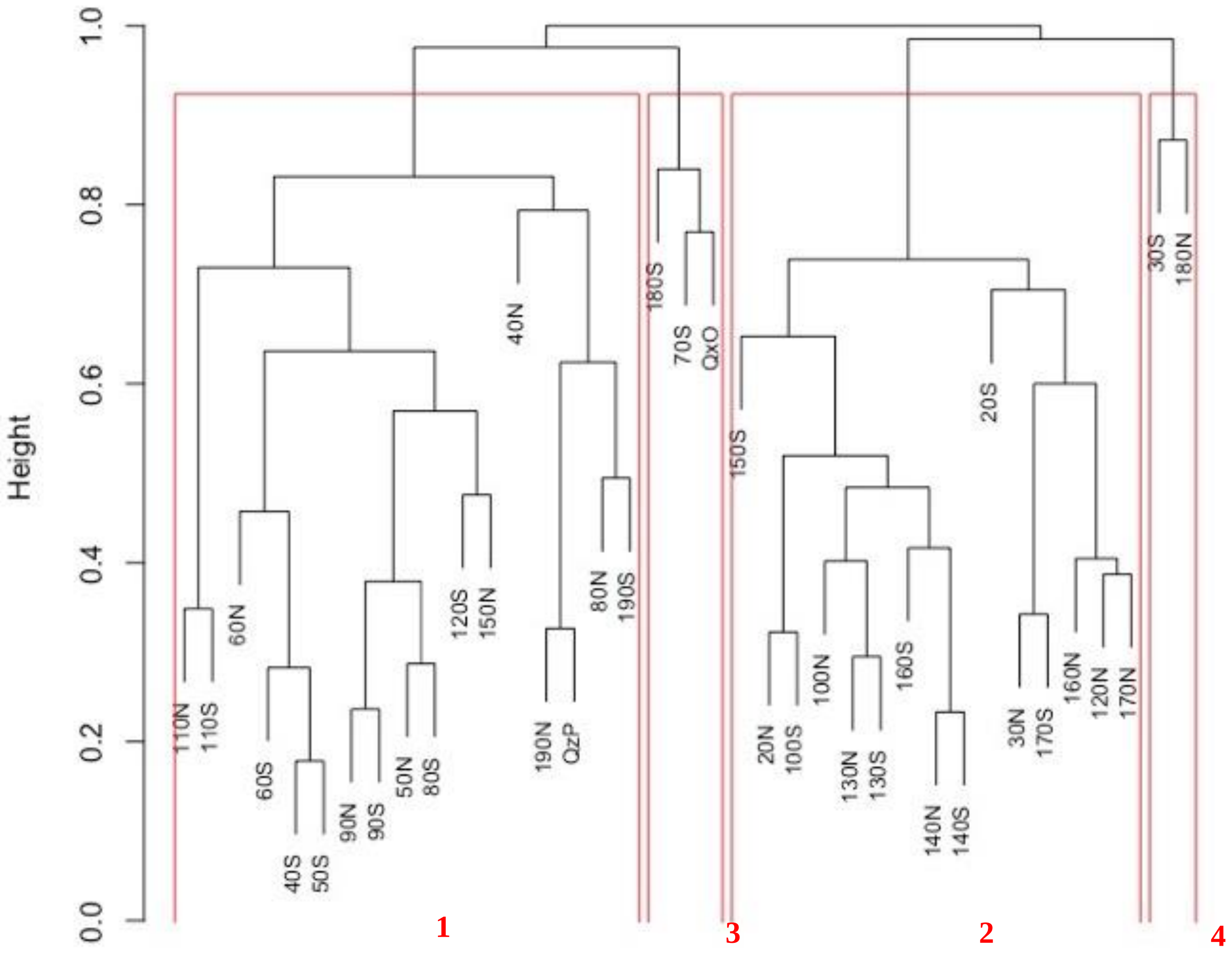


Figure 6: *Hierarchical cluster dendrogram of photo sites based on species abundance using the Bray-Curtis similarity measure. Red numbers and lines indicate groups of sites.*

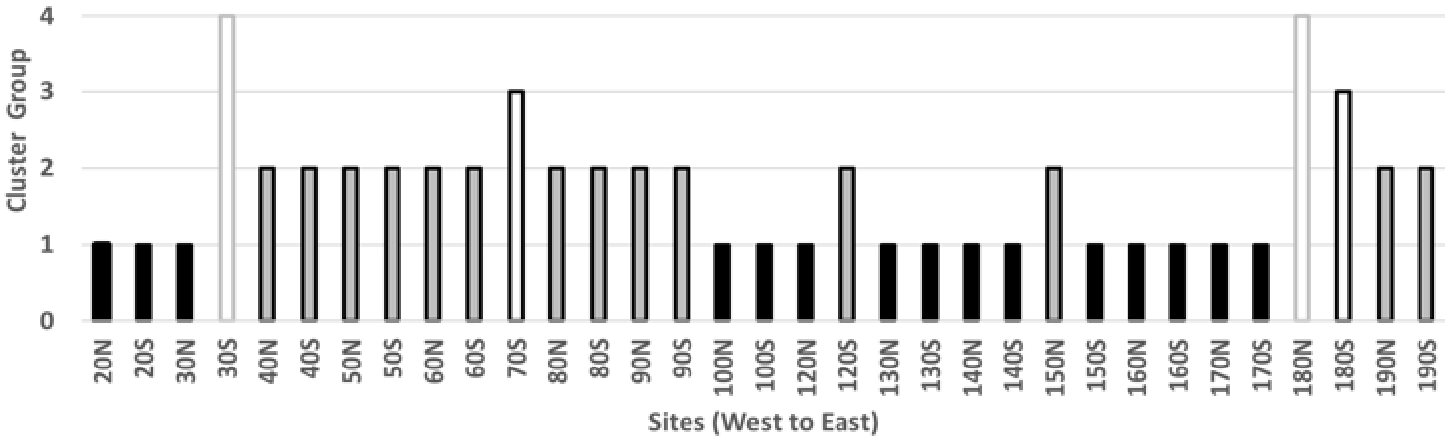


Figure 7*: Sites (x-axis) according to cluster group (y-axis) ordered from west to east. Black bars indicate group one, grey group two and white groups three and four.*

Cluster group one is dominated by seven species: *Dialium englerianum*, *Burkea africana*, Geoxylic suffruticosa (*Parinari capensis*) or *Diospyros lycioides*, *Strychnos pungens* and *Terminalia sericea*. Cluster group 2 is distinct from group one due to it having only three dominant species: *Baphia massaiensis*, *T. sericea* and *Baikea plurijuga*. *T. sericea* and *B. plurijuga* are at least twice as abundant in group two when compared to group one. Group three and four are clearly distinct from the grouping of species in group one and two due to the dominant presence of *Acacia nigrescens*, *Erythrophleum africanum*, *Gymnosporia senegalensis*, *Philenoptera nelsii*, *Philenoptera violacea* and *Bobgunnia madagascariensis*.

Table 3: The relative cover that each species comprises within its respective cluster group. Each cluster group contains a total species cover of 100%. For a comprehensive list of all species found at sites, see Appendix item 1C.

	Cluster Group (% Cover)			
Species	**1**	**2**	**3**	**4**
Acacia nigrescens	0.0	<1	38.4	6.8
Acacia erioloba	0.0	<1	4.3	6.3
Acacia hebeclada	0.0	0.0	0.0	1.6
Asparagus species	0.0	3.1	0.0	0.0
Baikea plurijuga	6.1	12.9	0.0	3.1
Baphia massaiensis	<1	23.4	0.0	1.6
Baissea wulfhorstii	0.0	2.1	0.0	0.0
Bauhinia urbaniana	5.3	3.7	0.0	0.0
Boscia albitrunca	0.0	1.5	7.2	0.0
Burkea africana	16.3	<1	0.0	0.0
Combretum collinum	<1	1.6	0.0	0.0
Combretum hereroense	1.3	0.0	1.1	1.6
Combretum imberbie	0.0	0.0	0.0	9.9
Combretum species	<1	6.7	0.0	0.0
Dialium engleranum	19.7	<1	0.0	2.6
Diplorhyncus condylocarpon	2.7	1.3	0.0	0.0
Erhythropleum africanum	3.1	<1	12.3	19.4
Euclea divinorum	0.0	0.0	0.0	5.8
Geoxylic suffritices	11.5	<1	0.0	0.0
Guibourtia coleosperma	2.9	<1	0.0	4.7
Gymnosporia senegalensis	0.0	<1	12.0	0.0
Philenoptera nelsii	0.0	<1	4.7	15.7
Philenoptera violacea	0.0	<1	0.0	12.6
Pterocarpus angolensis	3.1	2.1	0.0	0.0
Ochna pulcra	9.9	<1	0.0	0.0
Strychnos pungens	2.1	<1	0.0	0.0
Bobgunnia madagascariensis	<1	1.7	14.9	0.0
Terminalia sericea	11.8	31.0	1.4	2.6
Ximenia americana	<1	1.4	1.1	1.6

1.2. Woody Cover Change: EVI corroborates Photo Cover Scores

The Pearson correlation co-efficient returned a significant relationship between cover change calculated by EVI and photo cover scores (Figure 8: $R^2 = 0.67$; $p=<0.001$). This significant correlation suggests that there is agreement between two contrasting methods of cover estimation. Thus, results of woody cover change found using repeat photo cover scoring methods are corroborated by the EVI.

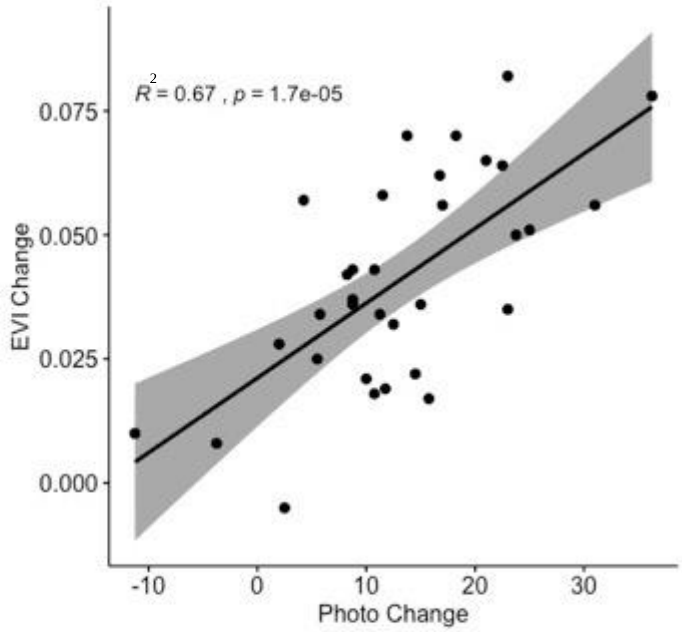


Figure 8: *Pearson correlation co-efficient with change in woody cover calculated by photo estimates (x-axis) and EVI (y-axis) plotted against each other. R^2 and p-value is listed in the top left corner. The grey area around line of fit indicates the 99% confidence interval and dispersion of data with black dots indicating raw data points.*

Objective 2: Describing inter-decadal scale patterns in rainfall, fire and land use across repeat photography sites.

2.1. CHIRPS Significantly Correlates with Observed Rainfall

The Pearson correlation co-efficient using CHIRPS and meteorological station rainfall datasets from Katima Mulilo (Figure 9A) and Rundu (Figure 9B) are highly correlated (Katima Mulilo/A: $R^2=0.93$ and $p=<0.001$; Rundu/B: $R^2=0.88$ and $p=<0.001$) (Figure 9) . Thus, the high-resolution CHIRPS rainfall data used in the GLMM can be considered as a reliable representation of the locally observed rainfall.

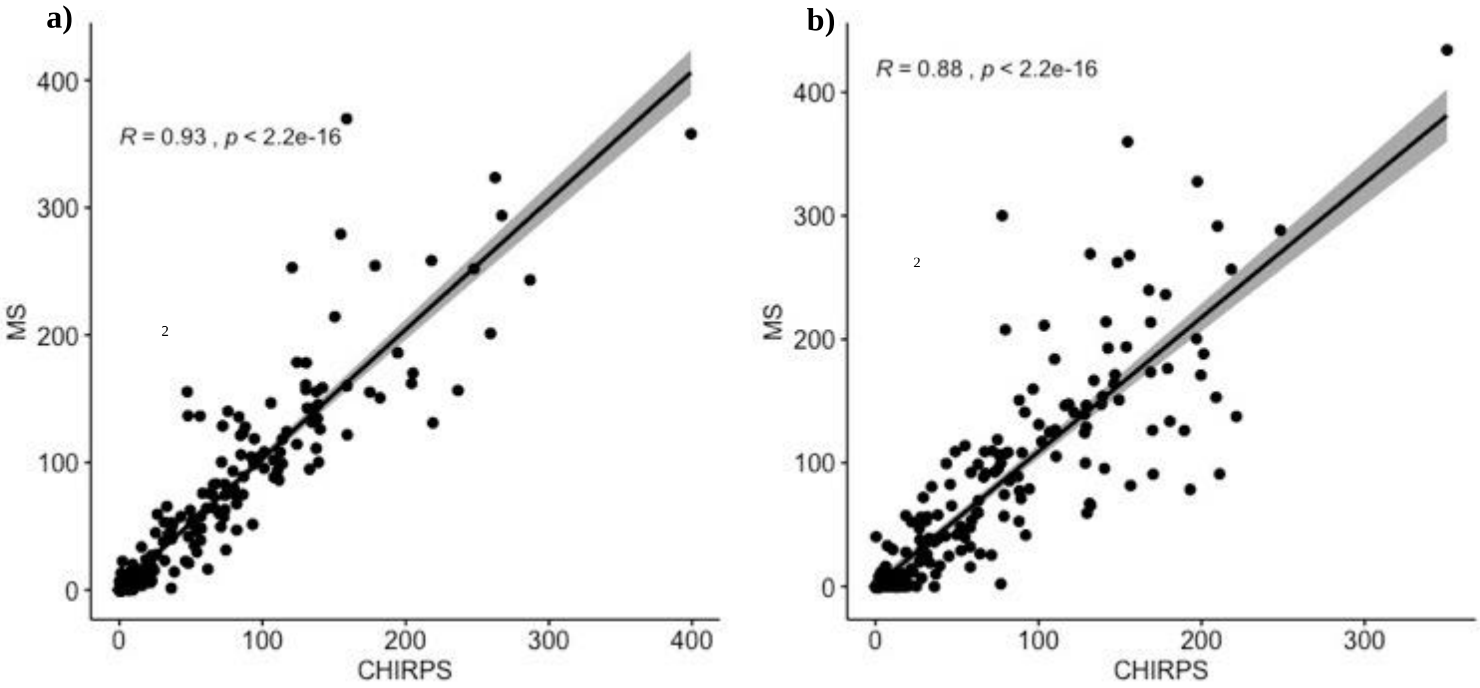


__Figure 9:__ Pearson correlation showing the Climate Hazards Group InfraRed Precipitation with Station data (CHIRPS) against observed rainfall from Namibian Meteorological Stations (MS) situated at Katima Mulilo (a; 110kms east of Bwabwata National Park) and Rundu (b; 180kms west of Bwabwata National Park) (www.meteona.com). R_2 and p value are listed in the top left hand corners.

2.2 CHIRPS: Antecedent rainfall to photo years

Photo year ‘18/’19 received 300mm less rainfall than photo year ‘98/’99 and has the lowest recorded rainfall since the ‘96/’97 rainfall season (Figure 10). Both photo years are below the MAP of BNP for the period ‘96/’97 – ‘18/’19 with ‘18/’19 being a drought year (www.meteona.com). The two year antecedent rainfall total for photo year ‘18/’19 is 920 mm and for photo year ‘98/’99 is 1010 mm (Figure 10. Considering the difference (90mm) in antecedent rainfall to photo years (920 and 1010 mm), the influence of rainfall on woody vegetation change is likely not significant.

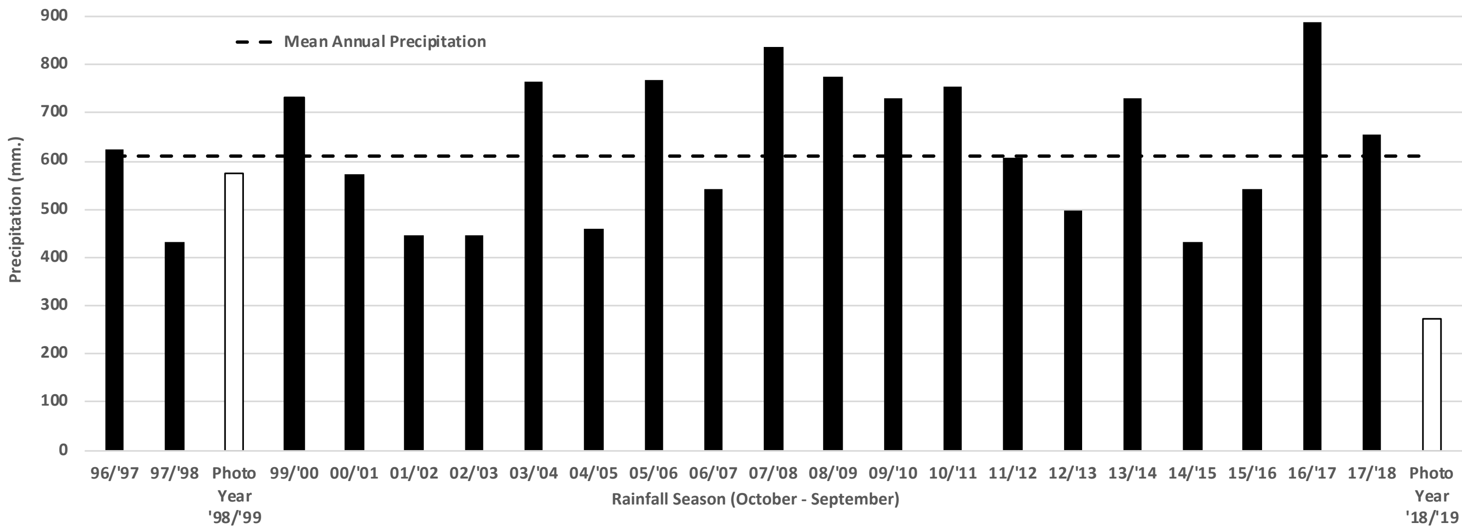


Figure 10: *Climate Hazards Group InfraRed Precipitation with Station data (CHIRPS) derived total annual rainfall for Bwabwata National Park from October – September rainy season '96/'97 – '18/'19. The '98/'99 and '18/'19 season when photos were taken are marked with white bars. The Mean Annual Precipitation (MAP) is marked with a dashed line in order to place antecedent rainfall of photo years into context.*

2.3. CHIRPS: West to East Rainfall Gradient

The MAP for the period '96/'97 – '18/'19 for pairs of sites displays a gradient of increasing rainfall from west to east (Figure 8). The largest range in MAP is 95mm and occurs between pairs of sites 100 km (minimum MAP) and 160km (maximum MAP). The average MAP for the period '96/'97 – '18/'19 of all sites is 593mm which is marginally lower than the MAP for the entire BNP (Figure 10 & 11).

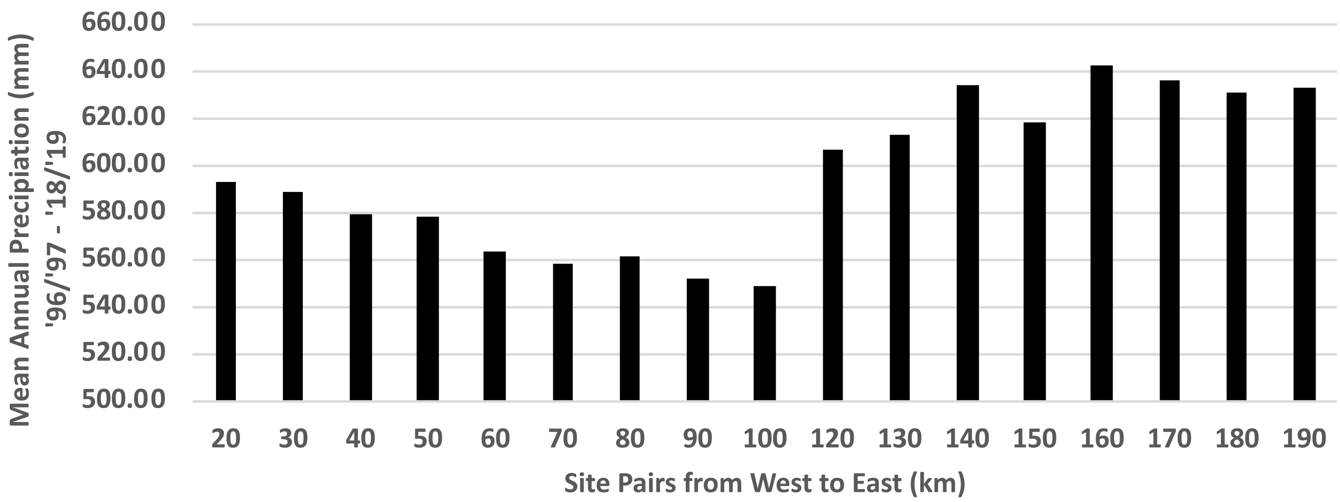


Figure 11*:Climate Hazards Group InfraRed Precipitation with Station data (CHIRPS) derived Mean Annual Precipitation (MAP; mm) for each pair of sites for the period '96/'97 – '18/'19. Sites (x-axis) are shown in pairs from west to east of Bwabwata National.*

2.4. MODIS Burned Area: Fire Return Interval and Seasonality of Burnt Area

The average FRI for all sites is seven years with sites in the MUA East displaying above average FRI and the majority in MUA West displaying below average FRI (Figure 12A). Nine (occurring within MUA West, CCA Buffalo and CCA Kwando) of 33 (27%) sites had a FRI of 1. Therefore, on average, a fire returned to nine of the study sites every year of the 18 years analysed. There were no sites located in the MUA East that contained such a comparably short FRI. The shortest FRI in MUA East was nine years and the longest was 18 years, which occurred in the MUA East. The average Late Dry Season Burnt Area (LDSBA) for all sites is 26% for the period of 2000 – 2018. Sites within the eastern portion of the park (MUA East and Kwando CCA) display below average LDSBA compared to most sites in the west of the park where LDSBA is above average. The average Early Dry Season Burnt Area (EDSBA) is much lower than LDSBA at only 8% of sites (Figure 12B).

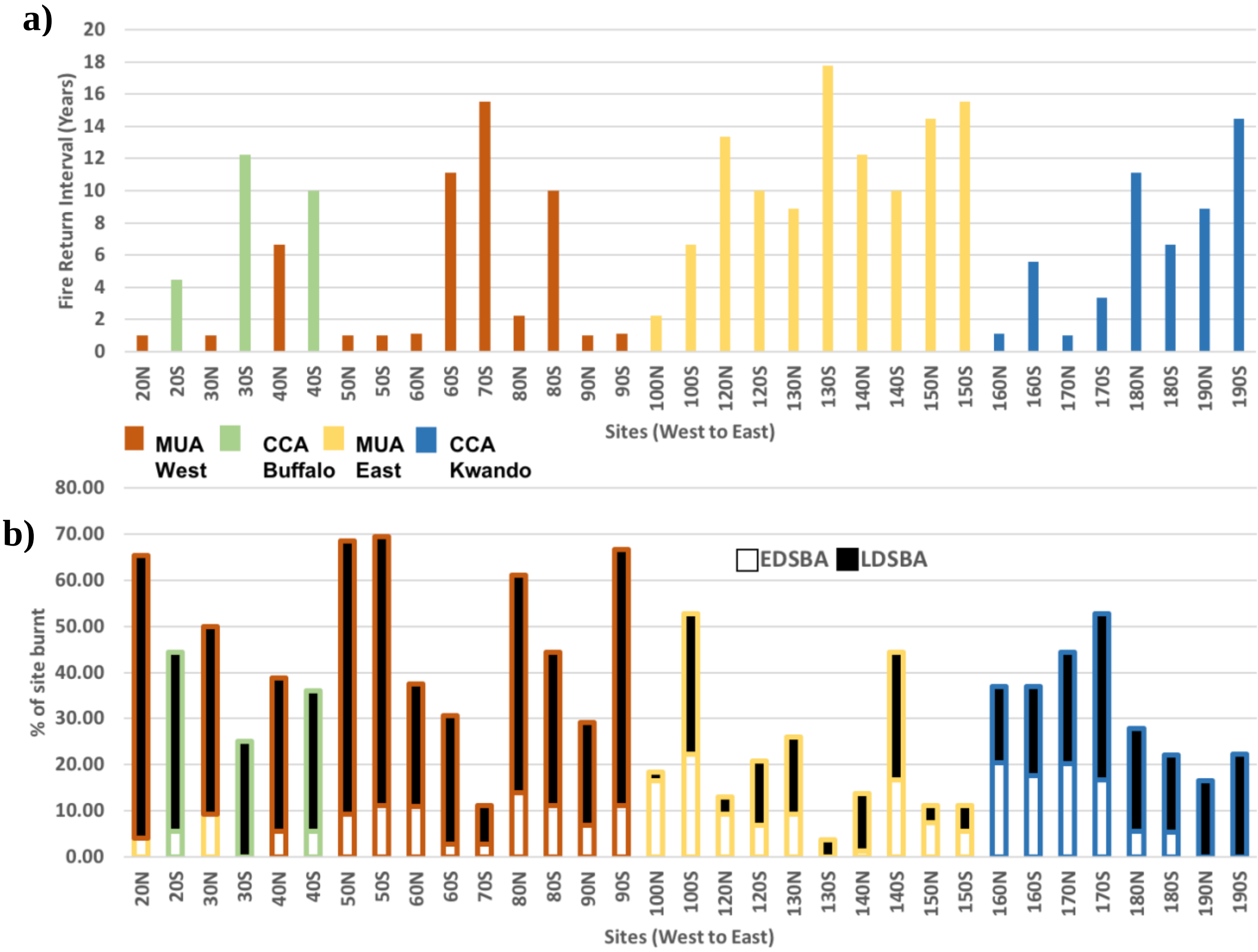


Figure 12: *Fire Return Interval (FRI) (A) and burn seasonality (B) derived from MODIS burned area product for 2000-2018. Land use categories are shaded to visualise different fire dynamics between areas. FRI in A identifies how many years it takes for a fire to return to a site. The burn seasonality in B differentiates the area of a site that burnt in the late dry season (LDSBA) and early dry season (EDSBA) ,and the total of the two bars represents the average percentage of burned area.*

2.5. Fire Management Differs Between Land Use Areas

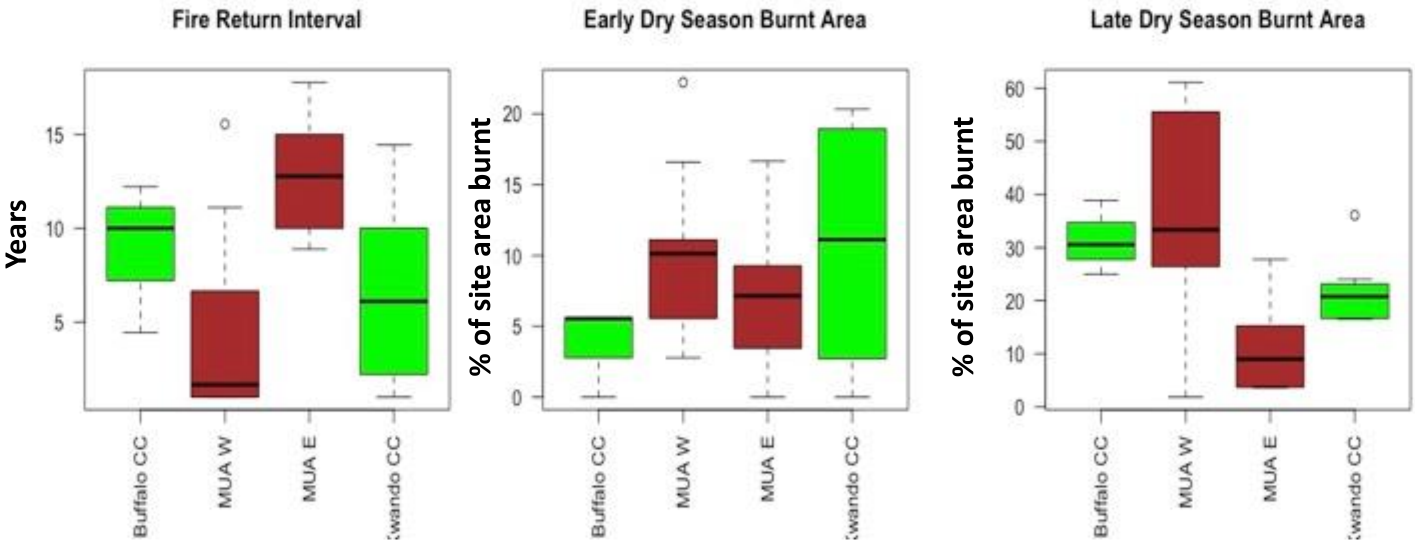


Figure 13: *Box and whisker diagrams of the three fire regime variables (FRI, Early Dry Season Burnt Area, Late Dry Season Burnt Area) derived from MODIS Burned Area product (2000 – 2018), grouped by the BNP land use categories (CCA Buffalo, MUA West, MUA East and CCA Kwando). Fire Return Interval displays the number of years it takes for a fire to return to a site. Early Dry Season Burnt Area displays the percentage of a site which burns during the early dry season (April – July). Late Dry Season Burnt Area displays the percentage of a site which burns during the late dry season (August – October). The CCA protected areas in BNP are highlighted in green and the MUA East and West in red.*

A one-way ANOVA revealed that all three fire variables differed significantly among land use categories (FRI: $F=38.86$, $p=<0.001$; EDSBA: $F=5.35$, $p=<0.001$; LDSBA: $F=30.17$, $p=<0.001$). The mean and upper and lower quartile values for each land use category are represented in the Box and Whisker plots (Figure 13). Tukey HSD results provided further detail in terms of which land use categories differed from each other (Table 4). The largest differences between land use categories were found for LDSBA where all comparisons except those involving CCA Buffalo (most likely due to a low sample size) were significant. Of the LDSBA comparisons, the most extreme was the difference between MUA West and MUA East where a highly significant difference of 27% was found. Differences between land use categories for EDSBA were not significant except between CCA Kwando and Buffalo and between MUA East and CCA Kwando. FRI was significantly different between all land use categories except between the CCAs Kwando and Buffalo. The largest difference (9 years) in FRI between land use areas was between the MUA West and MUA East. These results suggest that there is a significant difference between land use areas in terms of fire management application and that this difference is most evident between the categories of MUA West and MUA East.

Table 4: Results of Tukey Honest Significance of Difference test comparing fire variables across land use categories. Fire variables tested were derived from the MODIS Burned Area product and are Fire Return Interval (FRI), Early Dry Season Burnt Area (EDSBA) and Late Dry Season Burnt Area (LDSBA). Differences of the mean with a 95% confidence interval is provided as well as the p-value for each fire variable across all possible land use comparisons.

Land Use Comparison	Fire Variable	Mean of differences (95% CI)	p-value
CCA Kwando-CCA Buffalo	**FRI**	-1 (-5 to 3)	NS
	EDSBA	7 (1 to 13)	<0.05
	LDSBA	-12 (-26 to 1)	NS
MUA E-CCA Buffalo	**FRI**	6 (2 to 10)	<0.001
	EDSBA	1 (-5 to 7)	NS
	LDSBA	-23 (-37 to -10)	<0.001
MUA W-CCA Buffalo	**FRI**	-4 (-7 to 0)	<0.1
	EDSBA	4 (-1 to 10)	NS
	LDSBA	16 (8 to 24)	NS
MUA E- CCA Kwando	**FRI**	7 (4 - 10)	<0.001
	EDSBA	-6 (-10 to 1)	<0.01
	LDSBA	-11 (-20 to -2)	<0.05
MUA W-CCA Kwando	**FRI**	-2 (-5 to 0)	<0.1
	EDSBA	-3 (-6 to 1)	NS
	LDSBA	16 (8 to 24)	<0.001
MUA W-MUA E	**FRI**	-9 (-11 to -7)	<0.001
	EDSBA	-3 (-1 to 7)	NS
	LDSBA	27 (19 to 35)	<0.001

* NS – Non-significant

Objective 3: Investigating how fire seasonality, frequency, land use and rainfall drive woody cover change and structure.

3.1. Effect of Rainfall and Land-Use on Woody Vegetation Cover

Land use areas and MAP were significant determinants of change in all three woody cover categories (Table 6). The only land use area which did not display significant correlation with any woody cover change was CCA Buffalo due to a low sample size (Table 2; p = NS; df= 2). MUA West displayed the most significant correlation and effect with change in all woody cover categories. The eastern land use areas of MUA East and CCA Kwando displayed significant correlation with < 3 m and total woody cover change but not > 3 m woody cover change. Increasing MAP was correlated with an increase of > 3 m woody cover and total woody cover, but correlated with decreased woody cover in the < 3 m size class (Figure 14).

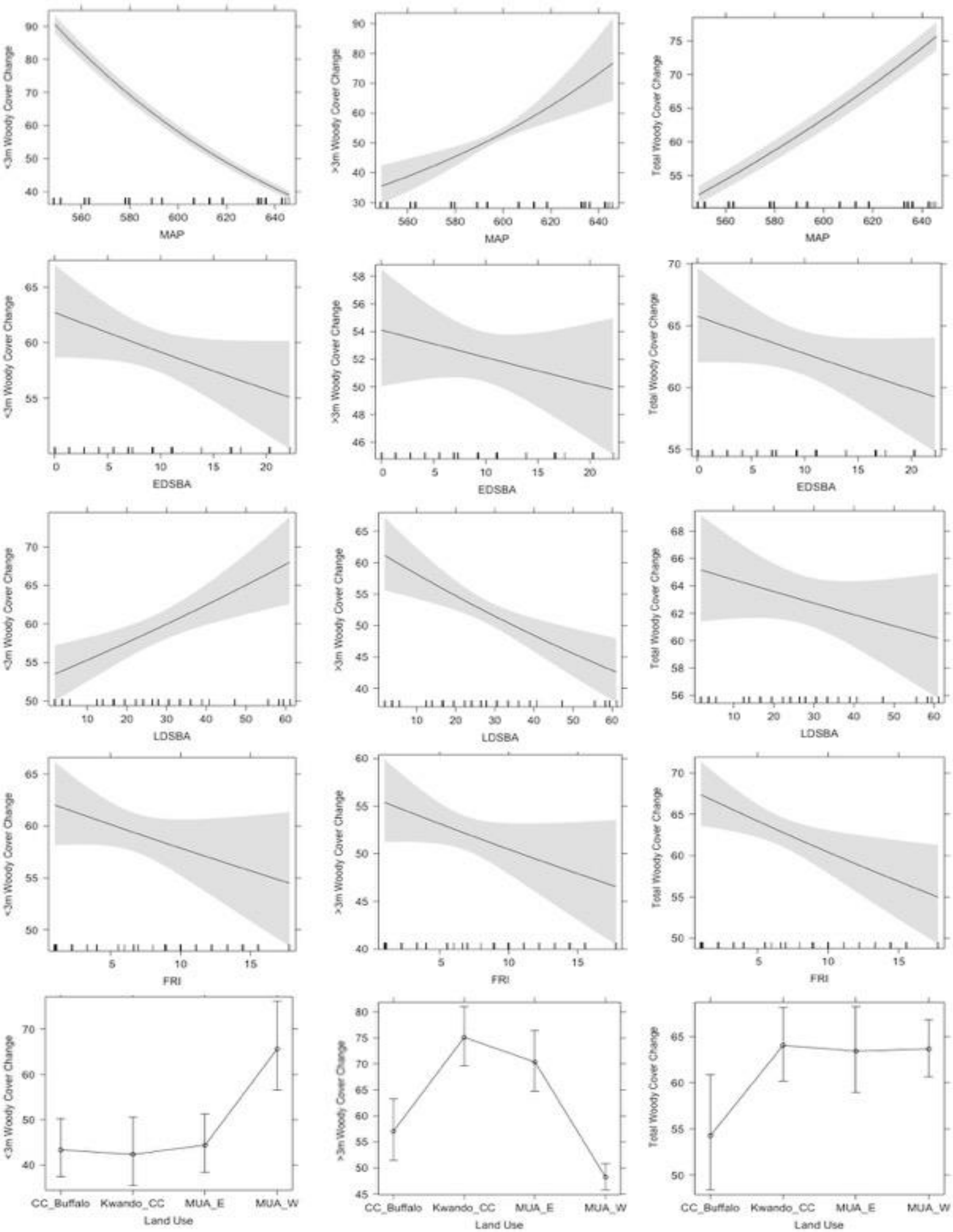
3.2. Effect of Fire Seasonality (LDSBA and EDSBA) and Frequency (FRI) on Woody Cover

LDSBA was a significant determinant of < 3 m woody cover change and > 3 m woody cover change. As photo sites increased in LDSBA, the change in < 3 m woody cover increased but change in >3 m woody cover decreased. While total woody cover was not significantly correlated with LDSBA, the relationship was negative (Figure 14, Table 6).

Total woody cover and < 3 m woody cover had a negative and significant correlation with EDSBA (Table 6). As the portion of a site burnt in the early dry season increased, the change of <3 m and total woody cover decreased (Figure 14). EDSBA was not significantly correlated with >3 m change but also displayed a negative relationship. FRI correlated significantly with total and >3 m woody cover. As FRI increased change in all woody cover categories decreased.

Table 5*: Summary statistics from the Generalised Linear Mixed Model, estimate of effect and associated standard error, t-value and p-value (NS- non-significant). All values for explanatory variables of Mean Annual Precipitation, Early Dry Season Burnt Area, Late Dry Season Burnt Area, Fire Return Interval and Land use categories with adequate sample size were included, however CCA Buffalo: n=3) was excluded.*

Response Variable	Explanatory Variable	Estimate	Standard Error	T value	P-value
< 3 m Woody Cover	MAP	-0.008	0.000	-144.400	<0.001
	EDSBA	-0.005	0.003	-1.824	<0.1
	LDSBA	0.004	0.001	3.477	<0.001
	FRI	-0.007	0.005	1.503	NS
	MUA West	-0.168	0.056	-2.982	<0.01
	MUA East	0.210	0.068	3.105	<0.01
	CCA Kwando	0.275	0.066	4.178	<0.001
> 3 m Woody Cover	MAP	0.007	0.001	4.309	<0.001
	EDSBA	-0.003	0.003	-1.011	NS
	LDSBA	-0.006	0.001	-3.539	<0.001
	FRI	0.010	0.006	-1.672	<0.1
	MUA West	0.415	0.082	5.008	<0.001
	MUA East	0.023	0.121	0.196	NS
	CCA Kwando	-0.022	0.139	-0.166	NS
Total Woody Cover	MAP	-0.001	0.000	-29.262	<0.001
	EDSBA	-0.004	0.002	-1.666	<0.1
	LDSBA	-0.001	0.001	-1.283	NS
	FRI	-0.012	0.004	-29.262	<0.001
	MUA West	0.160	0.061	2.611	<0.01
	MUA East	0.156	0.069	2.248	<0.05
	CCA Kwando	0.166	0.067	2.481	<0.05



<3m Woody Cover Change
>3m Woody Cover Change
Total Woody Cover Change
MAP
EDSBA
LDSBA
FRI
Land Use
CC_Buffalo
Kwando_CC
MUA_E
MUA_W

Figure 14: *Fitted regression functions of predicted effects derived from the Generalised Linear Mixed Model with untransformed data and slope angle indicating direction of change. Each explanatory variable (x-axis), Mean Annual Precipitation, Early Dry Season Burnt Area (EDSBA), Late Dry Season Burnt Area (LDSBA), Fire Return Interval (FRI) and Land Use category is arranged per row. Predicted effects are plotted for each response variable (y-axis; % change), < 3 m Woody Cover Change, > 3 m Woody Cover Change and Total Woody Cover Change in columns regardless of size or significance of effect. Y axis scales may differ according to the range of response observed for each category of woody cover change. The size of grey area around slope lines indicates the accuracy of fit for predicted effects. Dispersion of raw data is indicated in upper and lower quartiles for explanatory variable of land use - for the rest of the explanatory variables, the x-axis contains short vertical lines with thicker lines indicating a density of raw data points.*

CHAPTER 4: DISCUSSION

Objective 1: Bush Encroachment and Homogenisation of Vegetation Structure.

The results show that woody vegetation cover (canopy cover) in BNP increased by a total of 13% over the period 1999 – 2019, a rate of increase of 0.5% per annum, which is consistent with other findings in southern African savannas (O'Connor et al. 2014; Skowno et al. 2017; Stevens et al. 2016). majority of encroachment occurred within < 3 m woody cover, comprising 9% of the total 13% increase, suggesting a change in vegetation structure towards shrubs rather than large trees. The encroachment was not drastic enough to warrant a transition of the BNP savanna to a wooded savanna (Staver et al. 2011; Scholes 2003), however, evidence for the beginning of this transition was found as sites with total woody cover ≥ 50% increased from 4 in 1999 to 22 in 2019 (total sites = 33). Furthermore, there were no sites in 1999 which contained ≥50% cover of > 3 m woody vegetation but in 2019, there were two. These sites occurred in the east (MUA East and CCA Kwando) of the park.

Change in vegetation structure differed spatially across the park. The west of the park (CCA Buffalo and MUA West) increased significantly in < 3 m woody cover while the east increased significantly in > 3 m woody cover indicating that there is potentially a transition of savanna vegetation occurring. If all remains equal, it could be possible that the west of the park would transition into a shrub savanna and the east a wooded savanna (Scholes & Archer 1997).

In addition to the encroachment, homogenising of height structure (increased cover of woody vegetation <3m) has occurred predominantly in the west when compared to the eastern land use areas. Considering the links of vegetation structure heterogeneity to bird species richness (Sirami et al. 2009), this homogenising pattern in western land use areas might be considered a degradation of habitat (Benton et al. 2003).

For both land use areas of the west, >3 m woody cover declined by 3.5% and <3 m woody cover increased by 12.5%. In 1999 >3 m woody cover was 11.5% higher than <3 m but in 2019, this has switched around and there is 4.5% more <3 m than >3 m. The dominant increase of < 3 m woody cover in the western land use areas indicates that there is currently a high fuel load within the flame zone of grass and ground layer fires (van Wilgen et al. 2014; Govender et al. 2014). An increase in woody vegetation is associated with the shading out of the grass layer with less grass available to start fires (Hoffmann et al. 2005). However, in the case of reduced grass fuel load, there would likely be a concomitant increase in dry and dead woody vegetation on the ground layer of a similarly flammable property to that of grass (van Wilgen et al. 2014; Govender et al. 2014).

Cluster analysis based on the vegetation survey of woody species abundance revealed two distinct communities grouped in the east and west (Figure 7). The first community, situated predominantly in the east (10/15 sites in MUA East/CCA Kwando, 2/15 sites in MUA West, 3/15 sites in CCA Buffalo) was not dominated by any one particular species (Table 5). According to Mendelsohn & Roberts (1997), MUA East is unique in BNP in that it is dominated by Burkea-Teak Woodland (Figure 1) which might have influenced the community analysis results. The second community, grouped predominantly in the west of the park (10/14 sites in MUA West/CCA Buffalo, 2/14 sites in MUA East and 2/14 sites in CCA Kwando), was dominated by *T. sericea* and *B. massaiensis*. Both species have been observed to coppice and resprout at higher rates than other woody species following disturbance such as fire (Nefabas & Gambiza J 2007; Holdo 2005). These two species comprised over half of the woody cover in this community. While the sample method cannot be considered rigorous enough to justify extrapolating this trend in community to the park as a whole, it does warrant investigating *T. sericea* and *B. massaiensis* as indigenous encroachers. This would require repeating methods in this study over time to discern whether the two species are in fact increasing disproportionately more than others.

Comparison of repeat photography derived woody cover change and EVI derived productivity change between 1999 and 2019 returned significantly high correlation (R_2=0.67; $p<0.001$). Though repeat photography is sometimes criticised as a qualitative or subjective analysis and not as replicable as remote sensing based change studies (Webb et al. 2010), the results show that repeat photography can in fact return reliable woody cover change estimates (as also shown by Rohde et al. 2019; Hoffman et al. 2010 and others).

In summary, overall woody vegetation cover increased throughout the park in the study area, but the change in vegetation structure was heterogenous, with a trend towards increasing shrubby vegetation (<3 m) in the west and an increase in large trees in the east. The reasons and implications for these differences will be explored in the following sections.

Objective 2: Spatial and Temporal Patterns of Rainfall and Fire across BNP.

High resolution information on fire and rainfall over the study period were derived from MODIS C6 and CHIRPS datasets. Information on land-use was drawn from fieldwork (Example in Appendix 1C), management reports (MET 2013; 2016) and other studies in the area (Humphrey 2018; Taylor 2009 & 2012; Phaksi and Pyhälä 2018). This allowed for the analyses of variability of rainfall and fire experienced among sites to be analysed in combination with the land use and woody cover data. Rainfall as derived from the CHIRPS dataset correlated significantly with observed rainfall experienced at the nearest weather stations to BNP (Katima Mulilo: R_2=0.93 and p=<0.001; Rundu: R_2=0.88 and p=<0.001). Thus, it is concluded that the CHIRPS dataset can reliably be used to determine rainfall variables in areas where weather station data is not available. Unfortunately, the MODIS derived fire seasonality and frequency does not have a similar set of observed data to compare against. To test the reliability of MODIS fire information would require an independent study involving field work and data collection where ground truthing of fire events and their characteristics could be verified.

Analysis of CHIRPS derived MAP for each pair of photo sites revealed a decreasing gradient of rainfall from east to west of BNP. This is partially consistent with the report of Mendelsohn & Roberts (1997) that suggested a gradient of decreasing rainfall from north to south of the KAZA region but considering the NE – SW orientation of the park, this would imply a subtle gradient of decreasing rainfall from east to west. Thus, this study has confirmed that on average, in the order of 40mm more rainfall per annum occurs in the east of the park than the west (Figure 11).

MODIS data was used to reconstruct fire return interval (FRI), late dry season burn area (LDSBA) and early dry season burnt area (EDSBA) in different land-use areas across the park over the study period. Results from this study show significant differences in FRI and LDSBA between MUA West and MUA East (Figure 13). Most likely due to a low sample size in the two CCAs (3 photo sites in CCA Buffalo; 8 sites in CCA Kwando), there were no significant differences found between these land use areas. MUA West (13 photos sites) sites contained a significantly higher average LDSBA (34% of photo sites) compared to MUA East (12 photo sites) (7% of photo sites). Due to the association of late season fires with high intensity fires (Govender et al. 2006), this result suggests that fires in MUA West are hotter and larger than fires in MUA East and thus can be considered as high risk and hard to control (Humphrey 2018; Govender et al. 2006; van Wilgen et al. 2014). Additionally, the average frequency of fires was significantly shorter for sites in MUA West (every 2 years) compared to MUA East (every 13 years). This (particularly the seasonality findings) supports comments made in Namibia's fire management strategy (MET 2016), which states that MUA East fire management is organised and consistent and thus is able to prevent large uncontrolled fires in line with objectives of BNP. These results suggest that fire in MUA West is being used for different purposes to fire in the MUA East. LDSBA fires are more efficient at clearing large areas of vegetation for cultivation which is reported as a predominant objective of fire use by the Mbukushu people predominantly residing in MUA West (MET 2013 & 2016; Humphrey 2018). EDS fires are reportedly used more by the Khwe in MUA East - these results support this observation due to the lower occurrence of LDSBA fires compared to MUA West. This studies' findings concur with current management reports of BNP which provide a general statement that fire management differs between MUA East and MUA West in BNP (MET 2013 & 2016). The results of this study, in combination with those of Humphrey (2018) are now able to provide more insight into what exactly this difference in fire management is and as elaborated in Objective 1 and 3, how this alters woody vegetation in different ways.

The significant difference in seasonality between the MUA West and MUA East corresponds with a recent study by Humphrey (2018) in BNP. However, the fire return interval results contrast with Humphrey (2018) findings, who through analysis of a larger study area found that the MUA East contained the highest frequency of fire events in the park. This contrast in results could suggest that fire frequency is affected by proximity to the road as Humphrey (2018) conducted fire analyses throughout the park and not just along the roadsides which is a key difference between the two studies. Additionally, LDS fires are more prevalent along road sides due to their ability to spread far in the dry vegetation. However, LDS fires are not able to spread if an area is predominantly burnt in the EDS, as the fuel load connectivity is lower from the pattern of mosaic patch burns (Humphrey 2018). A possible explanation for this contrast in FRI results between Humphrey (2018) and this study could be that ignitions (almost all of which are anthropogenic) are likely concentrated along the roadside thereby altering FRI.

Objective 3: Prominent Late Dry Season Burning and Lower Rainfall Promotes Indigenous Encroachers.

To explore the drivers of increasing woody vegetation cover, a generalized mixed linear model (GLMM) with rainfall and fire characteristics was used to test which of the variables correlated most significantly with vegetation change. Comparison was performed with averages of variables as derived over the study period (1999 – 2019). The use of GLMM's as opposed to a GLM, allows the effect of site position on vegetation change to be tested and accounted for. This was necessary for the design of this study as the photo sites are too close to one another (400ms, see Figure 3) to be considered independent.

Results of the model suggest that rainfall (MAP), fire frequency (FRI) and seasonality (LDSBA and to some extent EDSBA) are primary drivers of woody vegetation in BNP considering their highly significant correlation with woody cover change (Table 6 and Figure 14). Total woody cover change had a positive relationship with MAP and a negative relationship with LDSBA, EDSBA and FRI.

As fires become less frequent at photo sites, the increase of woody cover in all categories decreased. This highlights how the presence of fire is a primary driver of woody vegetation change as in it's absence change was reduced. However, even though change was reduced by a reduction in fire frequency (long FRI), there was still an increase in woody vegetation, it was just less than sites with a higher FRI. This supports the idea that fire suppression can result in an increase, thickening or encroachment of woody species as found by O'Connor et al. (2014) and others. Furthermore, this result highlights that it is vital to understand not only how often fire is present but also the seasonality of fire in order to determine what effect the driver is having on vegetation change.

The divergence of height structure observed for the east and west of the park can be explained by the contrasting effects LDSBA has on < 3 m and > 3 m woody cover. The effect of LDSBA was positive for < 3 m woody cover and negative for > 3 m woody cover (Figure 14) across the park which concurs with findings in similar systems (eg Smit et al. 2010; Higgins et al. 2007; Smit et al. 2016). Considering that LDSBA was significantly higher in MUA west (Figure 13), it can be concluded that the lack of active fire management (i.e. no deliberate ignition of early season burns) is most likely responsible for prevalence of late season fires in this area (MET 2016) thus causing the increasing encroachment of < 3 m woody cover. In turn, this suggests that where early season prescribed burning is taking place, it is having a significant effect.

Due to the dominance of *B. massaiensis* (described as a predominantly <3 m shrub by Curtis & Mannheimer (2005)) and *T. sericea* (67% of distribution in Namibia is of various height categories (Curtis & Mannheimer 2005) in the western land use area, it is likely that these woody species are benefiting disproportionately from the high rate of LDSBA over other woody species. *B. massaiensis* is described as a pioneer species and it's increasing dominance of woody cover in MUA West could be interpreted as evidence that succession is not able to take place as a result of high disturbance from fire (Curtis & Mannheimer 2005). *T. sericea* is the 12_{th} most recorded species in Namibia (Curtis & Mannheimer 2005) and thus it's high cover in BNP is not out of pattern with other trends found in the country. These patterns support the listing of both *B. massaiensis* and *T. sericea* as indigenous invasive encroachers who benefit from frequent fire and can withstand drought conditions (Curtis & Mannheimer 2005).

The high significance of MAP from the GLMM results confirm traditional understanding of the influence of rainfall on woody cover growth (Sankaran et al. 2005; Figure 14). At sites with higher rainfall, woody cover is able to grow > 3 m at a quicker rate than at sites with less rainfall. Considering this, and the increasing gradient of rainfall from west to east in combination with high rates of LDSBA in the west, it is likely that tree cover > 3 m woody cover in the west will continue to decline. Unless fire management practice in the west is able to reduce the level of LDSBA occurring, the combined effects of lower rainfall and high LDSBA will continue contributing to what might be considered a positive feedback loop of high < 3 m woody cover, contributing towards the high level of LDSBA and preventing the growth of woody species to maturity/>3 m.

Total woody cover change and MAP displayed a significantly positive relationship (Table 6 and Figure 14; supporting traditional understanding of the relationship (Sankaran et al. 2005)), however, total woody cover across the park has increased despite low antecedent rainfall (Figure 10; Archibald et al. 2010). This provides initial evidence to suggest that global drivers (e.g. CO_2) or other factors not considered by this study such as herbivory may be influencing the woody cover change within BNP.

Thus, while there are global drivers arguably at play in woody cover encroachment across southern Africa, there are also local factors (i.e. fire management) which are in the hands of local authorities and communities which can be used to steer woody cover change into more desirable directions.

Objective 4: Management Implications and Conservation Concerns

In order to effectively achieve BNP's primary objectives of biodiversity preservation and sustainable economic development, an understanding of the state of woody vegetation and it's change is important. Woody vegetation forms an essential component of the services which an eco-system can provide in terms of biodiversity and livelihood preservation (Eldrige et al. 2011). This report has supported the idea that fire is a key determinant of the change in woody vegetation and thus the efficient implementation of the park's fire management can be considered as one of the most important actions which can be taken to achieve their primary goals.

One of the goals of BNP's fire management strategy is the reduction of wildfire risk, however, there are areas in the park where this risk is continuing at high levels (MET 2016). It was found that the risk of wildfires has increased and is disproportionately higher in the west than the east of BNP. If < 3 m woody cover is high, then the chance of fires spreading from ground level to > 3 m woody cover is increased as the < 3 m woody cover acts as a stepping stone for fire to reach higher levels (van Wilgen et al. 2014; Govender et al. 2014). This could create a positive feedback effect continuing the reduction of remaining > 3 m woody cover and homogenising of structure which is in turn interpreted as a degradation of habitat and contraventiton of the park's goal of biodiversity preservation (van Wilgen et al. 2014; Govender et al. 2014; MET 2013 & 2016). If, however, the western land use areas (CCA Buffalo and MUA West) were to increase their use of EDS mosaic patch burning, then it might be possible to reverse this trend. By burning frequently in the EDS they would reduce the grass fuel load and continuity of fuel creating patches of vegetation with different fire histories. Thus, if a LDS fire does occur the chance of it growing and spreading rapidly to > 3 m height would be decreased meaning < 3 m woody cover would be given time to grow > 3 m, allowing height structure to increase in heterogeneity.

The pattern of homogenising height structure in the west and increasing tree cover in the east is likely to have multiple effects on biodiversity, one of which might be the reduction in avian diversity (Sirami et al. 2009). However, with an increasing total woody cover across the park, there are likely also going to be reductions in grass and herb production if this study site conforms to analyses performed elsewhere (Scholes 2003; Smit 2004; Walker et al. 1972). Therefore, it is important for management to monitor how this encroachment has and will influence faunal abundance and movement. Considering the evident divergence of vegetation height structure between east (increase in both <3m and >3m woody cover) and west (disproportionate increase in <3m woody cover) of the park and the association between bird species richness and vegetation structure (Sirami et al. 2009; Benton et al. 2003), a comparison of avian diversity between areas might yield further insight into the effects of homogenising habitat and biodiversity. If in fact it is found that there is a reduction in avian diversity as a result of this change, there will likely be knock on consequences considering the important role that birds play in eco-system functioning particularly in savanna systems (Sirami et al. 2009) Additionally, it would be interesting to monitor the movement of grazers in relation to fire and encroachment patterns which promote or reduce the level of grass and herb production. It might be the case that particular fire management approaches are encouraging and others discouraging grazer movement. If fire management favours grazer presence disproportionately in one area over another there would be multiple impacts on other components of management concern such as predator movement, vegetation germination etc (Eldrige et al. 2011).

4.1. Study Limitations and Further Work

Due to the fact that this study involved a comparison of woody vegetation between 2019 and 1999 with no information on woody vegetation change in between, a potential criticism of the results is that despite the slower growth rates of woody vegetation as compared to grassy species (van Wilgen et al. 2014), perhaps the change found in 2019 has not been linear since 1999. If this were true, then it is likely that traditionally accepted drivers of woody vegetation such as antecedent rainfall (Archibald et al. 2010) would correspond to the change found (i.e. a decrease in woody cover found from 1999 to 2019 would correspond with lower rainfall in photo year 2019 than photo year 1999). However, considering the extremely low antecedent rainfall (Figure 10) of the latest photo year '18/'19 rainfall season compared to the first photo year '98/'99 rainfall season, a reduction in woody vegetation in the '18/'19 season would be expected, which is the opposite of this report's results. Thus, despite the low antecedent rainfall of the 2019 photo year there has still been an increase in woody vegetation, providing support for a linear change in woody vegetation since 1999.

This thesis shows how future vegetation change studies can utilise the strengths of both methods, providing insight into quantifiable change of large areas (remote sensing) which, if supplemented with selected repeat photography sites can be associated with structural and species composition change too. Considering this, it could be possible to increase the sample size area considered by this study by expanding the area of vegetation analysed by EVI. This would provide a greater area where total woody cover change has been derived from, therefore, improving confidence of total woody vegetation change results. If supplemented by selected repeat photo sites as in this study, then the total change derived from the broader EVI analyses could be tentatively associated with structural and species composition change. This method would require some initial methodological tests which would need to answer a few questions such as: how large an area needs to be covered by repeat photography analyses to be considered expandable into the area analysed by EVI? Is the area analysed by EVI representative of the area analysed by repeat photo sites?

Results of this study have identified how and why woody vegetation change is occurring and confirmed that EVI based measurement of change correlate to vegetation surveys on the ground. This highlights how the method could be upscaled to measure change in larger areas. The size of the study area cannot be considered as an accurate representation of the whole of BNP due to the small area relative to the total size of BNP that fieldwork was conducted within. The area of each photo site was less than 0.16 km^2. The total area surveyed covered 5.92km^2, which represented <1% of BNP's total area. Considering the trends found and the possible implications they would have on the goals of BNP, it is suggested that a larger vegetation survey and monitoring project be undertaken to provide further confidence in the initial results of this thesis. This could be achieved with relative ease if the use of remote sensing based change were to be extended across a larger area of the park. The high correlation of repeat photography and EVI based change found in this study suggests that an expansion of EVI based change would provide reliable results which could also be associated with height structure and species composition change.

For the purposes of this study, two reference years 1999 and 2019 were analysed. Explanatory variables are tested as averages (from 1999 - 2019), which doesn't account for the possible influence of variability within the study period (Rebertus et al. 1993). This is relevant for the analysis of FRI and might possibly explain the less significant influence of FRI on woody cover compared to LDSBA in this study (Figure 14). A site with a short FRI (eg. Figure 12 site 160N, 160S and 170S) can have high >3m woody cover due to the possibility that the first half of the study period ('99 – '09) contained minimal fire events and the last half ('09 - '19), fire events increased dramatically shortening the average FRI. In this case, the woody cover would be allowed to escape the fire trap in the first ten years of a long FRI and the short FRI average for the total period would not be able to provide insight into the influence that the FRI variability had on >3m woody cover (Rebertus et al. 1993; Hoffmann et al. 2012) Therefore, it is important to note when using this studies' results for estimating the influence of FRI on the ability of woody cover species to escape the fire trap, it is likely that they form an underestimate. This limitation, based on the time intervals of repeat photography monitoring projects highlights how analysis of satellite imagery can be used to create a more complete time series of change.

The need to further explore variability is also of concern for analysis of rainfall as MAP is considered as an over-exaggeration of rainfall due to the high levels of variability experienced from season to season in the study area (Mendelsohn & Roberts 1997). Considering the potential role of variation within explanatory variables (eg Rebertus et al. 1993), further analyses of the area or studies with similar objectives and questions would do well to unpack the variation within variables in addition to the averages.

While the impact of herbivory on post fire recovery of vegetation was not part of the scope of this study, this impact requires investigation. Domestic livestock and game herbivore density will decrease the rate at which vegetation is able to recover from fire (eg. Roques et al. 2016). MUA West contains a denser human population (MET 2013 & 2016) and thus it is possible that a subsequent concentration of domestic livestock (cows and goats) occurs here. Future research and monitoring should quantify livestock concentrations across BNP and assess the possible impact which it might have on woody species post fire recovery.

It is particularly important that this encroachment and the associated drivers are able to be placed within a longer temporal perspective of vegetation transitions within BNP (Gilson. 2009; Gillson & Willis. 2004). By gaining a longer temporal perspective of change, an idea of where ToC lie and whether it is possible to avoid them will be gained (Gillson & Duffin 2007). While initial attempts at gaining this longer perspective have been made (eg Dixon 2019), the question still remains of where this encroachment fits into the patterns of vegetation transition that BNP has experienced over a longer time period (centuries – millennia). This answer will provide perspective of the roles of global drivers such as $C0_2$ in the transitions of vegetation experienced within BNP so far. Once this is better understood it would be easier to motivate for the increased management of encroachment if in fact local drivers are primarily responsible. Or, in contrast, perhaps it would be understood that similar patterns of increased vegetation have occurred in the park before as a result of a global driver and there is little that local management can do to prevent it from occurring.

Finally, in this study, the influence of herbivory was not analyzed, nor were global drivers such as CO2 and nitrogen deposition. More complex models combing drivers that interact at different spatial and temporal scales were beyond the scope of this study but could be an interesting avenue of future research.

CHAPTER 5: CONCLUSION

This study has revealed how woody vegetation has changed in the last two decades and how it may be possible to manage this change with fire. Using repeat photography and satellite based remote sensing, there seems to be a general increase across the park, consistent with woody thickening and encroachment found in many other areas of the globe. This change was spatially disproportionate with an increase in < 3 m woody cover in the west of the park and an increase in > 3 m in the east. Woody species responsible for thickening in the west are listed indigenous invasives *B. massaiensis* and *T. sericea* occurring < 3 m and thus a degradation of habitat is considered to be occurring in the west.

A prominent gradient of decreasing rainfall from west to east was found for BNP with implications on fire fuel load. Despite the recent drought, woody vegetation has increased dramatically indicating that rainfall is not the only significant driver of change. When comparing fire seasonality and frequency between the management areas of BNP, significant differences were found between MUA West and East. Different use applications of fire by local communities can be considered to be driving the high portion of late dry season fires in the west and early dry season fires in the east. In analyzing how significant the different effects of rainfall, fire seasonality and frequency are in changing woody vegetation, it was possible to conclude that the primary reason for divergence of vegetation height in MUA West and East was seasonality of burning and rainfall gradient.

In the case of BNP, it seems that there is certainly a role that local fire management has in the change of woody vegetation. In order to reduce the encroachment of woody vegetation, late dry season fires can be used. However, frequent late dry season fires have been found to alter height structure and create a homogenous stand of indigenous invasives. Therefore, it is possible to argue that in fact EDS burning is advisable for BNP to adopt as a fire strategy considering their goal of biodiversity preservation and sustainable economic utilization (MET 2013; 2016). EDS burning prevents the buildup of a large area of continuous fuel load which encourages frequent LDS burns. These LDS burns can be considered responsible for the degradation of habitat in BNP, particularly in the western areas. In addition to preventing the degradation of habitat, EDS burning also increases wild food foraging opportunities (Humphrey 2018). EDS burning has been used as a livelihood and vegetation control method for thousands of years by the Khwe (Humphrey 2018) and contemporary evidence suggests the method is effective. Thus, the practice should be considered as a vital contribution towards the park's goal of biodiversity preservation and sustainable economic utilization.

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iv) Appendix

Appendix 1A, Photographic Scale:
2019 Woody Cover > 3m:

5% 	30% 
10% 	40% 
20% 	55% 

2019 Woody Cover < 3m:

5% 	30% 
10% 	50% 
20% 	60% 

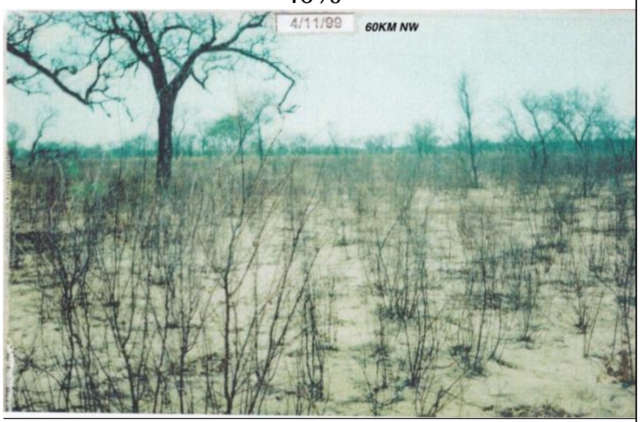
2019 Woody Cover Total:

<table>
<tr><td>20%
</td><td>50%
</td></tr>
<tr><td>30%
</td><td>60%
</td></tr>
<tr><td>40%
</td><td>70%
</td></tr>
</table>

1999 Woody cover > 3m:

5%	30%
	
10% 	40% 
20% 	

1999 Woody Cover < 3m:

5%  	30% 
10%  	40% 
20%  	

1999 Woody Cover Total:

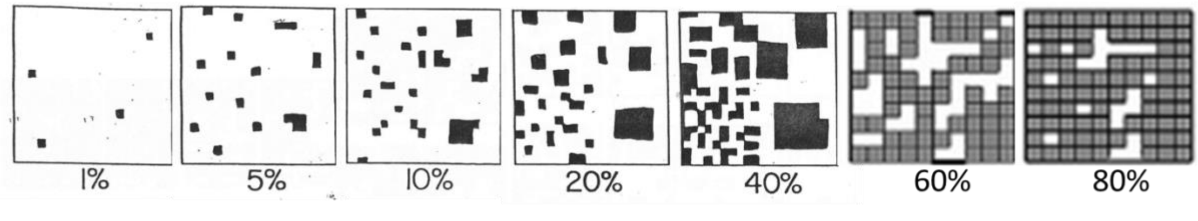
20%

40%

30%

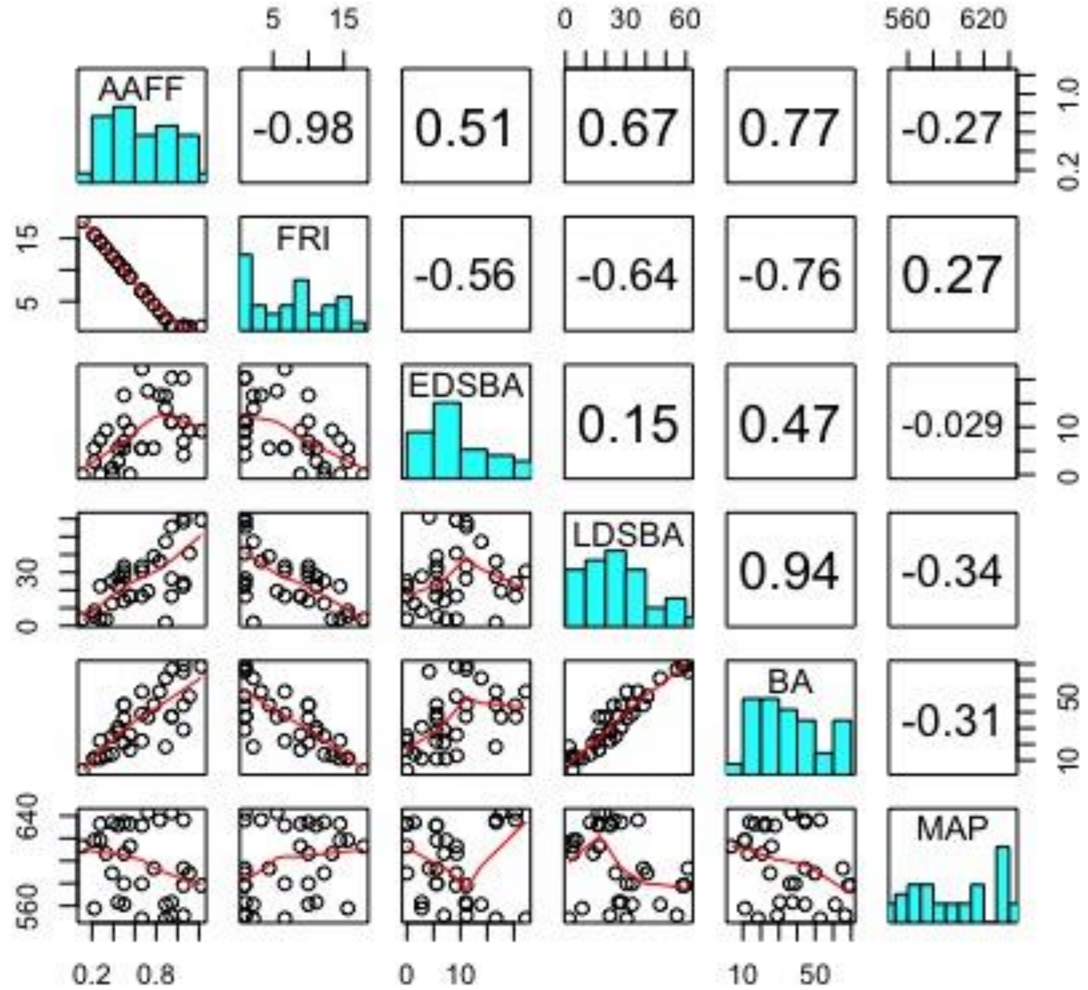
50%

Visual scale used for in field estimates:



Appendix 1B

6 explanatory variables (Annual Average Fire Frequency(AAFF); Fire Return Interval (FRI); Early Dry Season Burnt Area (EDSBA); Late Dry Season Burnt Area (LDSBA); Burnt Area (BA) and Mean Annual Precipitation (MAP)) were tested for their correlation. BA and AAFF were subsequently eliminated from analysis due to their correlation with FRI and LDSBA.



Appendix 1C:
List of all species found in photo sites across Bwabwata National Park with species authority included.

	Family	Species (*Genus species* Authority)
1	Fabaceae	*Acacia nigrescens* Oliv.
2	Fabaceae	*Acacia erioloba* E.Mey.
3	Fabaceae	*Acacia fleckii* Schinz
4	Fabaceae	*Acacia hebeclada* DC.
5	Fabaceae	*Acacia luederitzii* Engl.
6	Fabaceae	*Acacia* sp.
7	Fabaceae	*Albizia harveyi* E.Fourn.
8	Annonaceae	*Annona stenophylla* Engl. & Diels
9	Asparagaceae	*Asparagus* sp.
10	Fabaceae	*Baikea plurijuga* Harms
11	Fabaceae	*Baphia massaiensis* Taub.
12	Apocynaceae	*Baissea wulfhorstii* Schinz
13	Fabaceae	*Bauhinia urbaniana* Schinz
14	Capparaceae	*Boscia albitrunca* (Burch.) Gilg & Benedict
15	Fabaceae	*Burkea africana* Hook.
16	Rhamnaceae	*Berchemia discolor* (Klotzsch) Hemsl.
17	Combretaceae	*Combretum apiculatum* Sond.
18	Combretaceae	*Combretum collinum* Fresen.
19	Combretaceae	*Combretum hereroense* Schinz
20	Combretaceae	*Combretum imberbe* Wawra
21	Combretaceae	*Combretum psidioides* Welw.
22	Combretaceae	*Combretum* sp.
23	Combretaceae	*Combretum zeyheri* Sond.
24	Euphorbiaceae	*Croton gratissimus* Burch.
25	Fabaceae	*Dalbergia melanoxylon* Guill. & Perr.
26	Fabaceae	*Dialium englerianum* Henriq.
27	Fabaceae	*Dichrostachys cinerea* (L.) Wight & Arn.
28	Ebenaceae	*Diospyros lycioides* Desf.
29	Ebenaceae	*Diospyros virgata* (Gürke) Brenan
30	Ebenaceae	*Diospyros* sp.
31	Apocynaceae	*Diplorhynchus condylocarpon* (Müll.Arg.) Pichon
33	Fabaceae	*Erythrophleum africanum* (Benth.) Harms
34	Ebenaceae	*Euclea divinorum* Hiern
35	Fabaceae	*Guibourtia coleosperma* (Benth.) Leonard
36	Malvaceae	*Grewia avellana* Hiern
37	Malvaceae	*Grewia flava* DC.
38	Malvaceae	*Grewia* sp.
39	Celastraceae	*Gymnosporia senegalensis* (Lam.) Loes.
40	Bignoniaceae	*Markhamia obtusifolia* (Baker) Sprague

41	Rubiaceae	*Pavetta schumanniana* F.Hoffm. ex K.Schum.
42	Rubiaceae	*Pavetta zeyheri* Sond.
43	Rubiaceae	*Pavetta* sp.
44	Chrysobalanaceae	*Parinari capensis* Harv.
45	Fabaceae	*Peltophorum africanum* Sond.
47	Fabaceae	*Philenoptera nelsii* (Schinz) Schrire
48	Fabaceae	*Philenoptera violacea* (Klotzsch) Schrire
49	Fabaceae	*Pterocarpus angolensis* DC.
50	Phyllanthaceae	*Pseudolachnostylis maprouneifolia* Pax
51	Ochnaceae	*Ochna pulchra* Hook.
52	Euphorbiaceae	*Schinziophyton rautanenii* (Schinz) Radcl.-Sm.
53	Anacardiaceae	*Searsia marlothii* (Engl.) Moffett
54	Loganiaceae	*Strychnos cocculoides* Baker
55	Loganiaceae	*Strychnos pungens* Soler.
56	Fabaceae	*Bobgunnia madagascariensis* (Desv.) J.H.Kirkbr. & Wiersema
57	Combretaceae	*Terminalia sericea* Burch. ex DC.
58	Fabaceae	Fabaceae sp.
59	Rubiaceae	Rubiaceae sp.
60	Lamiaceae	*Vitex mombassae* Vatke
61	Lamiaceae	*Vitex* sp.
62	Olacaceae	*Ximenia americana* L.
63	Olacaceae	*Ximenia caffra* Sond.
64	Rhamnaceae	*Ziziphus mucronata* Willd.

Full species list and relative cover according to cluster grouping:

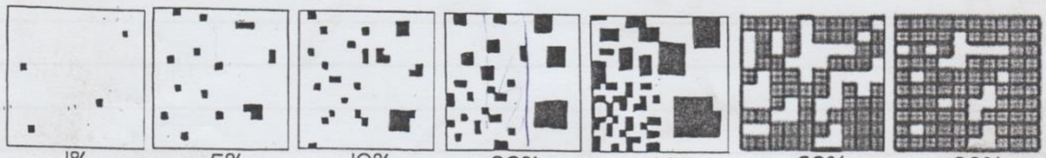
Group	1.0	2.0	3.0	4.0
Acacia nigrescens	0.0	<1	38.4	6.8
Acacia erioloba	0.0	<1	4.3	6.3
Acacia fleckii	0.0	<1	0.0	0.0
Acacia hebeclada	0.0	0.0	0.0	1.6
Acacia luederitzii	0.0	<1	0.0	0.0
Acacia species	0.0	<1	0.0	0.0
Albizia harveyi	0.0	0.0	<1	1.0
Annona stenophylla	<1	0.0	0.0	0.0
Asparagus species	0.0	3.1	0.0	0.0
Baikea plurijuga	6.1	12.9	0.0	3.1
Baphia massaiensis	<1	23.4	0.0	1.6
Baissea wulfhorstii	0.0	2.1	0.0	0.0
Bauhinia urbaniana	5.3	3.7	0.0	0.0
Boscia albitrunca	0.0	1.5	7.2	0.0
Burkea africana	16.3	<1	0.0	0.0
Combretum apiculatum	<1	0.0	0.0	0.0

Combretum collinum	<1	1.6	0.0	0.0
Combretum hereroense	1.3	0.0	1.1	1.6
Combretum imburbie	0.0	0.0	0.0	9.9
Combretum psidioides	<1	0.0	0.0	0.0
Combretum species	<1	6.7	0.0	0.0
Combretum zeyheri	0.0	<1	0.0	0.0
Dalbergia melanoxylon	<1	<1	0.0	0.0
Dialium engleranum	19.7	<1	0.0	2.6
Diospyros lycoides	0.0	0.0	<1	<1
Diospyros virgatta	<1	0.0	0.0	0.0
Diplorhyncus condylocarpon	2.7	1.3	0.0	0.0
Erhythropleum africanum	3.1	<1	12.3	19.4
Euclea divinorum	0.0	0.0	0.0	5.8
Geophylic suffritices	11.5	<1	0.0	0.0
Guibourtia coleosperma	2.9	<1	0.0	4.7
Grewia avellane	0.0	<1	0.0	0.0
Grewia flava	0.0	<1	0.0	0.0
Grewia species	<1	<1	<1	0.0
Gymnosporia senegalensis	0.0	<1	12.0	0.0
Markhamia obtusifolia	0.0	0.0	0.0	0.0
Pavetta schumania	<1	<1	0.0	0.0
Pavetta zeyheri	<1	0.0	0.0	0.0
Peltophorum africanum	0.0	0.0	0.0	1.0
Philenoptera nelsii	0.0	<1	4.7	15.7
Philenoptera violacea	0.0	<1	0.0	12.6
Pterocarpus angolensis	3.1	2.1	0.0	0.0
Pseuodlachnostylis maprouneifolia	<1	0.0	0.0	0.0
Ochna pulcra	9.9	<1	0.0	0.0
Schinziophyton rautanenii	<1	0.0	0.0	0.0
Searsia marlothii	0.0	0.0	0.0	<1
Strychnos cocculoides	<1	0.0	0.0	0.0
Strychnos pungens	2.1	<1	0.0	0.0
Swartzia madagascariensis	<1	1.7	14.9	0.0
Terminalia sericea	11.8	31.0	1.4	2.6
Unknown Fabacea	0.0	<1	0.0	0.0
Unknown Rubiaceae	<1	0.0	<1	<1
Vitex mombassa	0.0	<1	0.0	0.0
Ximenia americana	<1	1.4	1.1	1.6
Ximenia caffra	<1	<1	<1	<1

Appendix 1D: Example of Data Sheet Used in field for vegetation survey and photo methods

Site no: 1105/1882 **DETAILED ECOLOGICAL DESCRIPTION** Date: 4/10/'19

No.	Species name	% cover N	% cover E	% cover S	% cover W	No.	Species name	% cover N	% cover E	% cover S	% cover W
1	Comp. sp.	3	3	3	4	26					
2	Ac. eri	6	0	0	3	27					
3	Ter. ser.	10	3	5	3	28					
4	Bap. mas.	3	5	6	2	29					
5	Bau. urb.	2	3	2	3	30					
6	Bai. plu.	1	5	3	4	31					
7	Xym. caf.	0	1	0	1	32					
8	Geo sup.	<1	1	0	0	33					
9	Oich. cin.	0	0	<1	0	34					
10	Och. pul.	0	0	<1	<1	35					
11	Grew. fl.	0	0	0	<1	36					
12						37					
13						38					
14						39					
15						40					
16						41					
17						42					
18						43					
19						44					
20						45					
21						46					
22						47					
23						48					
24						49					
25						50					
Bare ground Cover		50	20	25	35	**Woody Cover (suffrutices)**		<1	<1	0	0
Rock/Stone Cover		0	0	0	0	**Woody Cover (<3 m)**		10	20	20	10
Grass Cover		<1	<1	<1	<1	**Woody Cover (>3 m)**		15	12	12	10



1% 5% 10% 20% 40% 60% 80%

Site no: 1105 / 1882 **GENERAL ECOLOGICAL DESCRIPTION** **Date:** 04/10/'19

Landscape description relative to topographical features (e.g. base of dune; directly next to omuramba, sandy plain, etc.) Sandy plain, N. of omuramba.

Soil type (circle one): sand / loamy sand / sandy loam / loam / sandy clay loam / sandy clay / clay

Erosion: Wind or water.

0 (no visible impact)	2	3	4	5 (recent major impact affecting most of view)

Vegetation Type (circle one): Burkea Shrubland / Burkea-Combretum Woodland / Burkea-Kiaat False Mopane / Burkea-Teak Woodland / Burkea-Terminalia Woodland / Okavango-Kwando Valley Woodland / Omuramba Fringe Woodland / Omuramba Grassland / Open Camelthorn Woodland / Teak Savanna / Teak Shrubland / Teak Woodland / Other But closer to teak woodland.

Dominant growth forms and species: Ter. ser. (1-8m.); Bai. (1-8); Bap-mas. (1-3m.); Comp. sp. (1-8m.); & Bau. urb. (1-3m.).

Herbivory: browsing or grazing, broken branches from elephants, other observations
No herbivory but heavy human dist.

Fire history (circle one): recently burnt / charred stumps of trees / branches indicating high intensity fires / browning of the leaves / other Burnt in June 2019.

Description of major changes: Seasonal change:
major reduction in grasscover; BG increase
Total woody cover seems similar
Woody cover > 3m. likely lower due to loss of multiple tall trees (atleast 6 > 3m. trees from felling + fire).
~~[illegible]~~ Possible increase in WC < 3m. due to frequent fire but difficult to say due to picture repeat seasonality.

Site no: 1105/1882 **NOTES ON SPECIFIC REGIONS/INDIVIDUALS** **Date:** 04/10/11

Development of a powerline in southern section of site, parallel to road. (trees cleared from site). Disturbed site situated on the eastern edge of Uetto (Perry's house). Abundance of heavy resprouting of shrubs under 3m indicates frequently burnt site (possibly early burning). No ground cover/grassland so may have been burnt in the late season. Site dominated by Combretum, Terminal and Acacia shrubs Baphia. Maaiveus, fav. Urbiana. Trees greater than 6m large understory (<3m shrubs).

GT1105 - Ac. Welsby / Small vaun >6m
GT1105 - 5m Comb spp.
GT1105 - B. plu >6m height surrounded by shrubs <3m.
GT1105 - T. ser (large tree >6m) next to village house - 6m B. plu next door
GT1105 - Comb spp >6m / comb. apic 5m / next door x2 (dense stand)
GT1105 - Comb Tm >6m (fallen down)
GT1105 - Open / shrubs - opengrassland / bare ground - Ter. ser / com. apic Bap. ma <3m in height.
GT1105 - 2x comb. Apic trees >6m height
GT1105 - Open area / Bare ground - late season fire evidence - shrubs less than 3m in height
GT1105 - T. ser >6m
GT1105 - B. plu >6m in height surrounded by comb. api trees / T. ser >6m trees.

Ter. ser. (1-8m.) 7

Small Bai. plu. (1m.) exist

SITE INFORMATION

Site No: 1105/ 1882 **Date:** 04/10/'19 **Time spent on site:** 1h 40 m

Concise Site Name: 1105

General description of location: S of road, E of small huts, nearby land for clearing.

Coordinates: S 17.95651 **Altitude (m):** 990m.

E 22.56734 **Waypoint #:** 1105C

Original Photographer: Robyn Beatty.

Original number: 1105 **Original date:** 03/2006

Notes on accuracy of repeat photo station location: Inaccurate, fallen trees.

Names of team members present on site: Glynis, Adele, Conor

PHOTOGRAPHIC INFORMATION **Photographer's Name:** Conor Easbment

Camera make & model: Canon 5D MKII **Camera Height:** 1m.

How has the site been marked? Rod removed; GPS point

Notes on weather conditions: Sunny, No clouds

Frame or IMG No.	Lens Focal Length	Exposure (Speed)	f-stop	Compass bearing (°)	Time
2429	24	125	8.0	N	8:33
2430	24	200	"	E	8:34
2431	24	125	"	S	8:35
2432	24	200	"	W	8:37
2433	40	~~40~~ 125	"	N	8:38
2434	40	~~40~~ 160	"	E	8:39
2435	40	~~40~~	"	S	8:40
2436	35	~~35~~	"	W	8:42

Appendix 2: Research Permit granted by Namibian National Commission on Research Science and Technology.

FORM RST/4

NATIONAL COMMISSION ON RESEARCH, SCIENCE AND TECHNOLOGY

RESEARCH, SCIENCE AND TECHNOLOGY ACT, 2004

RESEARCH PERMIT
FOR
NON-NAMIBIAN-BASED RESEARCH
INSTITUTE/PERSON

(Section 21 and Regulation 22)

Permit Number
RPIV00842019

Name of Non-Namibian-based Research Institute/Person: Associate Professor Lindsey Gillson Co-workers: Glynis Humphrey, Adele Julies, Conor Eastment, Imago Katanga, Timm Hoffmann and Sally Archibald.	**Physical Address:** HW Pearson Building, University Avenue North, Rondebosch, Cape Town, 7701 South Africa
Issue Date: 19 July 2019	**Commence Date:** 01 September 2019
Termination Date: 30 September 2020 renewable	**Sample Taking Authorised:** YES ✓ NO

Type of Research Authorised
Landscape history, biodiversity and fire management in Bwabwata National Park, north-eastern Namibia.

Non Commercial research and the use of the resources be limited to what is specified in the research proposal.

Type and Size of Sample Taking Authorised
Small (3cm3) surface samples of pollen will be collected from small water bodies and 100 photographs

Locations Authorised for Research and /or Sample Taking
Plant Conservation Unit, University of Cape Town;

Intended Use of Samples
For export for further research at the Plant Conservation Unit, University of Cape Town;

Responsible Person: Lindsey Gillson	**Contact No:** +27 21 650 5552

Signature Removed

Signed on behalf of the National Commission on Research, Science & Technology

Official Stamp of the National Commission on Research, Science & Technology